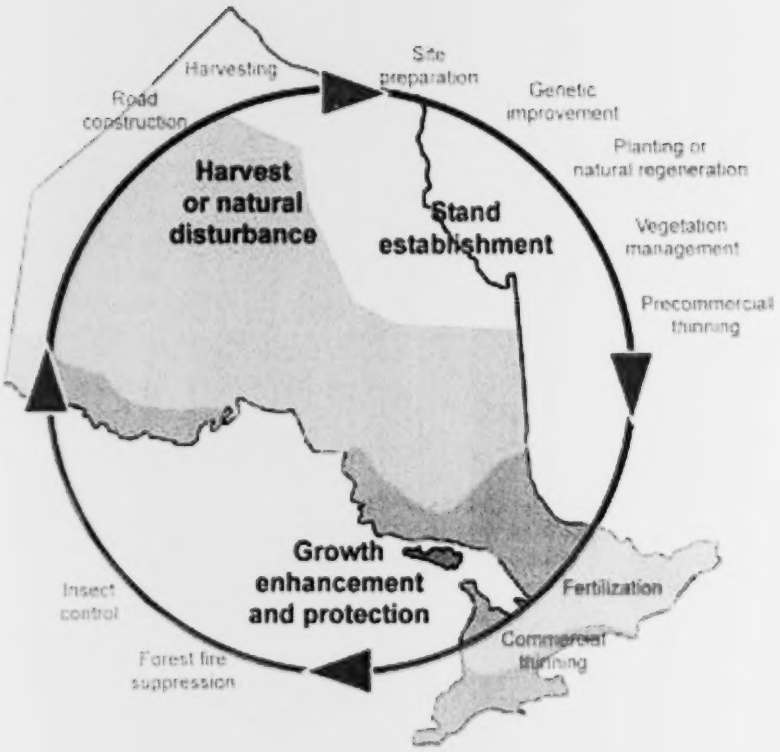
03

CLIMATE CHANGE RESEARCH REPORT

CCRR-03

The Effects of Forest Management on Carbon Storage in Ontario's Forests



Ontario

Symbol	Unit	Number of kilograms	Number of tonnes[1]	CO_2 equivalent[2] tonnes
g	gram	0.001	1×10^{-6}	3.67×10^{-6}
kg	kilogram	1	0.001	0.00367
t	tonne	1,000	1	3.67
kt	kilotonne	1 million	1 thousand	3,670
Mt	megatonne	1 billion	1 million	3.67 million

[1] The international system
yard-pound system om
multiples of the gram. Thus, a mass of 1000 kg (1 million grams) is called a tonne (t) instead of a megagram, and one million t is called a megatonne (Mt) instead of a terao
done here.

[2] The term CO_2 equivalent (CO_2eq) refers to the relative heat trapping abilities of different gases in the atmosphere. Gases released as a result of human activity, including CO_2, methane (CH_4), and hydrochlorofluorocarbons (HCFC), trap long-wave radiation from the sun in the atmosphere. The greenhouse effect of each gas depends on its molecular properties and the length of time the gas resides in the atmosphere. A CO_2 equivalent expresses the effect of different greenhouse gases relative to the greenhouse gas effect of the same amount of CO_2.

The Effects of Forest Management on Carbon Storage in Ontario's Forests

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Summary

This report examines how forest management can affect the carbon (C) balance of Ontario's forests. Ten forest management activities organized in four themes were examined: stand establishment (site preparation, planting, and vegetation management), growth enhancement (thinning, fertilization, and genetic improvement), forest protection (from forest fires, and insect and disease infestations), and harvesting (controlling the area occupied by roads, skid trails and landings, and reducing the area disturbed by harvesting).

The C storage values in this report represent preliminary estimates based on simplified assumptions about forest productivity and species composition. Estimates were obtained from the published literature and by developing a generalized forest soil, litter, and biomass scenario using stand yield data for Site Class 2 black spruce in Ontario. Stand yield data were input into version 2 of the Carbon Budget Model of the Canadian Forest Sector (CBM-CFS2), which has been used to model the C budgets of forests in Ontario and elsewhere in Canada. The effect of forest management practices on stand-level C was scaled to the provincial level using amount of area treated with each practice since 1990.

With the exception of thinning, all of the management practices examined can be used to increase forest C storage during a rotation. Here we summarize the effect of each forest management practice on changes in forest C storage focussing on the period from 1990 to 2012.

1. **Planting:** On average almost 73,000 ha were planted annually in Ontario from 1990 through 2000, ranging from about 56,000 to 93,000 ha. Extrapolating this average planting rate, the total C gain due to planting between 1990 and 2012 will be about 1.6 million t more than that obtained from natural regeneration.
2. **Site preparation:** Site preparation can affect C storage in forests both positively and negatively depending on intensity, that is, level of forest floor and mineral soil disturbance. Site preparation treatments can reduce C storage if they result in nutrient loss or increased forest floor or soil microbial respiration. In contrast, site preparation can increase C storage by reducing competition and promoting more rapid establishment of new stands, which increases the sequestration of C in living biomass and litter. Mechanical site preparation has had only minor direct effects on forest soil C levels, while prescribed burning reduces forest floor C. Chemical site preparation likely has the least influence on forest floor and soil C. However, with any form of site preparation, the long-term benefit of more rapid stand establishment increases C sequestration and more than compensates for initial losses of forest floor or soil C that may occur. Investigations of the net effect of site preparation on forest C are needed.
3. **Vegetation management:** Delayed or inadequate vegetation management (i.e., competition control) can significantly reduce crop tree growth in plantations and severe competition can cause significant mortality and conversion of stands to a barren and scattered condition. The forest C gains from planting versus those from vegetation management are inseparable, since these practices are usually applied to the same stands. The C gains accrued from planting as compared to natural regeneration, described above, therefore includes an estimate of the effect of vegetation management on plantation C storage.
4. **Thinning**: Stand density regulation (spacing, thinning) may increase yields of merchantable volume over a rotation, but does not increase total biomass production or C sequestration per unit land area. However, the rotation for sawlogs may be reduced significantly and stem quality improved, with a slight positive impact on C storage in longer-lived wood products. An indirect reduction in CO_2 emissions may occur where these larger stems replace energy-intensive structural materials such as steel, aluminum, and concrete, which generate large CO_2 emissions during their manufacture.

5. **Fertilization**: Fertilization increases tree biomass accumulation and C storage. In Ontario, jack pine is the best candidate for fertilization. A single application of N fertilizer to semi-mature, fully stocked jack pine stands would store an additional 4.9 t C per hectare over 10 years. However, greenhouse gases generated by inorganic fertilizer production and use could offset 5 to 10% of the C storage gain. The net forest C gain between 2008 and 2012 from fertilizing 10,000 ha of jack pine each year from 2005 through 2011 would total almost 164,000 t.
6. **Genetic improvement**: By 2012, the gain in biomass C from all genetically improved stands planted since 1990 would be over 76,000 t relative to planting non-improved stock.
7. **Forest fire management**: Forest fire management reduces the area burned and thus increases forest C storage. Practising intensive fire management in the current 'measured' fire management zone could reduce the area burned by over 20,000 ha in an 'average' fire year, and would retain an additional 4.1 million t of stored C in forests between 2008 and 2012.
8. **Insects and diseases**: Should a major spruce budworm outbreak occur, protecting forests by spraying insecticide would prevent growth losses and tree mortality. A 60,000 ha annual spray program of black spruce stands that began in 2008 would increase C sequestration by about 311,000 t C by 2012.
9. **Roads and landings**: Areas deforested to build roads or landings have less C than forest regenerated by planting after clearcut harvesting. The total C loss in roads and landings built since 1990 will be approximately 960,000 t C in 2012 compared to replanted forest. Reducing the area of new roads and landings in the managed forest by 10% beginning in 2005 would reduce C loss by about 9,000 t. Extending forest management north into virgin forest requires the construction of new access roads. If from 2005 to 2011, 10,000 ha of productive forest north of the Planning Area were harvested annually, the resulting deforestation due to roads and landings would reduce C storage by about 9,000 t.
10. **Disturbance (forest harvest)**: Reducing the annual harvest by 1,000 ha (a 0.5% reduction in harvest over the approximately 200,000 ha annually harvested), would reduce harvest in each of the Ontario's 47 forest management units by slightly more than 20 ha. This would retain about 190,000 t of biomass C annually from 2008 through 2012.

A significant change in forest C storage can reflect the sustainability of forest management. Increasing C storage not only helps mitigate atmospheric greenhouse gas accumulation and global climate change, but is also an indicator of increased ability of forests to support economically, biologically, and socially important activities. The practices examined in this report do not all produce large increases in forest C between 2008 and 2012, the first Kyoto accounting period, but over the long term almost always increase both C storage and wood supply.

Acknowledgements

Funding for this project was provided by the Interministerial Climate Change Committee under the auspices of project CC-112. Werner Kurz and Zhanxue Zhu of the Canadian Forest Service, Pacific Forestry Centre, provided the CBM-CFS2 simulation that was the basis for C estimates used throughout this report. We are also grateful to those who reviewed parts of this document. These include Werner Kurz; Colin Bowling, Eric Boysen, Bill Cole, Shelagh Duckett, Paul Gray, Pengxin Lu, Rob McAlpine, Jim McLaughlin, John McLaughlin, Andrée Morneault, Dave Morris and Cathy Nielsen of the Ontario Ministry of Natural Resources; Tim Lynham and Pierre Bernier of the Canadian Forest Service; Peter Higgelke of KBM Forestry Consultants, John McNutt of Tembec, and Barry Davidson of Westwind Forest Stewardship.

Résumé

Le présent rapport traite des incidences que les activités d'aménagement forestier peuvent avoir sur le bilan du carbone des forêts de l'Ontario. Dix activités d'aménagement organisées en quatre thèmes ont été examinées : l'établissement du peuplement (préparation du site, plantation, gestion de la végétation), l'amélioration de la croissance (éclaircie, fertilisation et amélioration génétique), la protection de la forêt (contre les feux et les infestations d'insectes et d'agents pathogènes) et l'exploitation (limitation des superficies occupées par les chemins, les pistes de débardage et les chantiers de façonnage, réduction des superficies perturbées par les travaux de récolte).

Les valeurs chiffrées concernant les stocks de carbone (C) qui paraissent dans le présent rapport sont des estimations préliminaires fondées sur des hypothèses simplifiées sur la productivité de la forêt et la composition des peuplements. Nous sommes parvenus à ces estimations à l'issue d'un dépouillement des écrits qui ont été publiés et de l'élaboration d'un scénario généralisé sur le sol, la litière et la biomasse de la forêt à partir des données sur le rendement d'un peuplement d'épinettes noires dans une station de classe 2. Les données sur le rendement du peuplement ont été intégrées à la version 2 du Modèle du bilan du carbone pour le secteur forestier canadien (CBM-CFS2) qui a servi à modéliser les bilans du carbone des forêts de l'Ontario et d'autres régions du Canada.

Hormis les éclaircies, toutes les pratiques d'aménagement forestier qui ont été examinées ici peuvent être utilisées pour accroître le stockage du carbone dans les forêts sur la durée d'une rotation. Nous résumons ci-après l'incidence de chaque pratique sur l'évolution des stocks de carbone des forêts pour la période 1990–2012.

1. **Préparation du site** : La préparation du site peut avoir des incidences tant avantageuses que défavorables sur le stockage du carbone dans les forêts selon l'intensité, ou l'ampleur, des perturbations subies par la couche holorganique et le sol minéral de la forêt. Les traitements de préparation du site peuvent réduire les stocks de carbone s'ils entraînent une perte d'éléments nutritifs ou s'ils accroissent la respiration de la flore microbienne de la couche holorganique ou du sol forestier. À l'inverse, certains traitements de préparation du site peuvent augmenter les stocks de carbone en réduisant la concurrence et en favorisant un établissement plus rapide du nouveau peuplement, ce qui augmente la séquestration du carbone dans la litière et la biomasse vivante. La préparation mécanique du site n'a eu que des incidences directes mineures sur les niveaux de carbone dans le sol forestier, alors que le brûlis prescrit a diminué le carbone dans la couche holorganique. La préparation chimique du site est probablement la méthode qui a le moins influé sur le carbone du sol forestier. Cependant, tous les modes de préparation du site procurent des avantages à long terme en accélérant l'établissement du peuplement et, donc, en favorisant une augmentation de la séquestration du carbone qui dépasse largement toutes les pertes de carbone qu'ils ont pu provoquer initialement dans le sol forestier ou la couche holorganique. Des recherches plus poussées s'imposent pour mesurer l'effet net de la préparation du site sur le carbone des forêts.
2. **Plantation** : En Ontario, de 1990 à 2000, près de 73 000 hectares ont été reboisés chaque année, la fourchette variant de 56 000 à 93 000 hectares. Une extrapolation de ce rythme de plantation annuel moyen autorise à prévoir que le gain de carbone total attribuable au reboisement effectué de 1990 à 2012 sera supérieur d'environ 1,6 million de tonnes à celui qui résulterait de la régénération naturelle.
3. **Gestion de la végétation** : Le retard ou l'insuffisance au chapitre de la gestion de la végétation (lutte contre les végétaux concurrents) peut réduire sensiblement la croissance des arbres de rapport; en cas de concurrence sévère, de nombreux arbres peuvent mourir, le peuplement peut s'éclaircir excessivement ou même disparaître par endroits et il ne faut plus guère compter que sur la régénération naturelle. L'augmentation des stocks de carbone des forêts attribuable à la plantation est inséparable de l'augmentation attribuable à la gestion de la végétation, puisque ces pratiques sont habituellement appliquées aux mêmes peuplements. Les gains en carbone attribuables à la plantation comparés aux gains en carbone attribuables à la régénération naturelle évoqués ci-dessus incluent par conséquent une estimation de l'effet de la gestion de la végétation sur le stockage de carbone.

4. **Éclaircies** : Les interventions destinées à réguler la densité du peuplement (distances de plantation, éclaircies) peuvent augmenter les rendements en produits commercialisables sur la durée d'une rotation, mais elles n'augmentent pas la production de biomasse totale ou la séquestration de carbone par unité de surface. Cependant, elles peuvent raccourcir la durée d'une rotation visant l'obtention de grumes de sciage et augmenter la qualité des tiges, avec une légère augmentation du stockage de carbone dans les produits ligneux à vie plus longue. Une réduction indirecte des émissions de CO2 peut être obtenue lorsque ces tiges de gros diamètre remplacent des matériaux de gros oeuvre comme l'acier, l'aluminium et le béton dont la fabrication est énergivore et rejette d'importantes quantités de CO2.

5. **Fertilisation** : La fertilisation augmente l'accumulation de la biomasse ligneuse et le stockage de carbone. En Ontario, le pin gris est le meilleur candidat à la fertilisation. Un seul épandage d'engrais azoté dans un peuplement dense de pins gris d'âge moyen permettrait d'augmenter de 4,9 tonnes à l'hectare le stockage de carbone au bout de dix ans. Cependant, il faudrait minorer ce gain de 5 à 10 % pour tenir compte des gaz à effet de serre générés par la fabrication et l'épandage d'un engrais inorganique. En fertilisant 10 000 ha de pins gris chaque année, de 2005 à 2011, le gain net qui serait obtenu de 2008 à 2012, en matière de stockage de carbone, se chiffrerait au total à près de 164 000 tonnes.

6. **Amélioration génétique** : D'ici à 2012, l'augmentation des stocks de carbone dans la biomasse produite par tous les peuplements établis depuis 1990 avec des sujets génétiquement améliorés serait de plus de 76 000 tonnes, comparativement aux peuplements constitués de sujets non améliorés.

7. **Lutte contre les feux de forêt** : La lutte contre les feux de forêts limite les superficies ravagées par le feu et, de ce fait, augmente le stockage de carbone dans la forêt. Les pratiques intensives de lutte contre les feux de forêts qui sont menées dans l'actuelle zone de gestion des feux « mesurée » pourraient réduire de plus de 20 000 hectares la superficie brûlée au cours d'une année de feu « moyenne », et donc retiendraient 4,1 millions de tonnes de carbone de plus dans les forêts de 2008 à 2012.

8. **Insectes et maladies** : Si un foyer d'infestation grave par la tordeuse des bourgeons de l'épinette se déclarait, la protection des forêts par pulvérisations insecticides préviendrait les pertes associées à la diminution de croissance et à la mort d'un certain pourcentage d'arbres. Un programme qui prévoierait la pulvérisation d'un insecticide sur 60 000 ha de pins noirs annuellement, à partir de 2008, ferait croître le stockage de carbone d'environ 311 000 tonnes d'ici à 2012.

9. **Chemins forestiers et chantiers de façonnage** : Dans les aires des forêts qui sont aménagées en chemins ou en chantiers de façonnage, le stockage de carbone est moins élevé que dans les aires qui ont été replantées après une coupe à blanc en 1990. La perte totale en carbone associée à la construction de nouveaux chemins et chantiers de façonnage depuis 1990 se chiffrera à quelque 960 000 tonnes en 2012, par comparaison avec les aires reboisées. Une réduction de 10 % de la superficie occupée par les chemins et les chantiers de façonnage construits à partir de 2005 réduirait la perte de carbone d'environ 9 000 tonnes. De nouveaux chemins d'accès devront être construits pour étendre l'exploitation forestière vers le nord dans la forêt vierge. Si, annuellement, de 2005 à 2011, on récoltait 10 000 hectares de forêt productive au nord de la zone de planification, le déboisement causé par la construction des chemins et des chantiers de façonnage réduirait les stocks de carbone d'environ 9 000 tonnes.

10. **Perturbation (exploitation de la forêt) :** Une réduction de la récolte annuelle de 1 000 ha (soit 0,5 % des 200 000 ha de forêt coupés annuellement) représenterait une diminution d'à peine plus de 20 ha de la surface récoltée dans chacune des 49 unités de gestion forestière de l'Ontario. Cette mesure permettrait de retenir annuellement environ 190 000 tonnes de carbone de la biomasse, de 2008 à 2012.

Une évolution significative du budget du carbone des forêts témoigne de la durabilité de l'aménagement forestier. Un stockage accru du carbone non seulement contribue à atténuer l'accumulation atmosphérique de gaz à effet de serre et le changement climatique global, mais il traduit aussi une capacité accrue des forêts à soutenir des activités importantes au plan économique, biologique et social. Les pratiques examinées dans le présent rapport n'aboutissent pas toutes à des augmentations importantes des stocks de carbone dans les forêts de 2008 à 2012, première période pour laquelle une comptabilisation du carbone sera exigée par le protocole de Kyoto, mais, à plus longue échéance, elles feront croître presque toujours à la fois le stockage de carbone et la production de bois.

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1. Introduction

1. Introduction

Forest Management, Climatic Change, and the Kyoto Protocol

There is international agreement that anthropogenic greenhouse gas (GHG) emissions need to be curbed to avoid dangerous changes in global climate (IPCC 2001). In response, Canada and more than 160 other countries have ratified the Kyoto Protocol to establish binding GHG emission reduction targets for industrialized nations. The Kyoto Protocol requires developed countries to reduce anthropogenic GHG (e.g., carbon dioxide (CO_2), methane (CH_4), nitrous oxide (NO_2)) emissions during the period 2008 to 2012 by an average of 5.2% below 1990 levels. Canada volunteered to reduce annual emissions by 6% below the 1990 level.

Greenhouse gas emissions in Canada totalled an estimated 607 megatonnes (Mt) of CO_2 equivalents (CO_2eq) in 1990 (1 Mt equals one million t or 1 billion kilograms[1]) (Fig. 1.1). Ontario's 1990 emissions accounted for 181 Mt of this total and along with Quebec, Alberta, and British Columbia, accounted for about 80% of the Canadian total. In 1990, about three quarters of Canada's and Ontario's GHG emissions came from fossil fuel combustion for energy generation and transportation[2]. Actual and projected growth in Canada's GHG emissions since 1990 (Fig. 1.1) indicates that the reduction needed to keep Canada's commitment to the Kyoto Protocol could be as much as 280 Mt CO_2eq yr^{-1} (Fig. 1.2).

Ontario, with its large population and industrial base, contributed about 30% of Canada's annual GHG emissions in 1990. In comparison, Alberta emitted about 28% of the Canadian total while Quebec and British Columbia accounted for about 14% and 9% of emissions, respectively. From 1990 to 2000, the growth in emission rates in Alberta was 52 Mt compared to 26 Mt in Ontario. If emissions continue to increase at these rates, by 2010 Alberta will be emitting 275 Mt and Ontario 233 Mt of GHG each year (Fig. 1.1). Based on these projected emissions, these four provinces would need to reduce GHG emissions by a combined total of 80% to meet Canada's 6% reduction target (Fig. 1.2). While Canadian provinces and territories will not be held individually responsible for meeting GHG emissions targets, it is anticipated that population centres and industrial sectors responsible for the bulk of the emissions will be most affected by efforts to achieve emission reductions. Thus, attention to curbing emissions and promoting sinks for GHG are in both the provincial and national interest.

[1] For conversion factors, see inside front cover.
[2] Data from Environment Canada's website (http://www.ec.gc.ca/pdb/ghg/1990_00_report/). Accessed November 17, 2003.

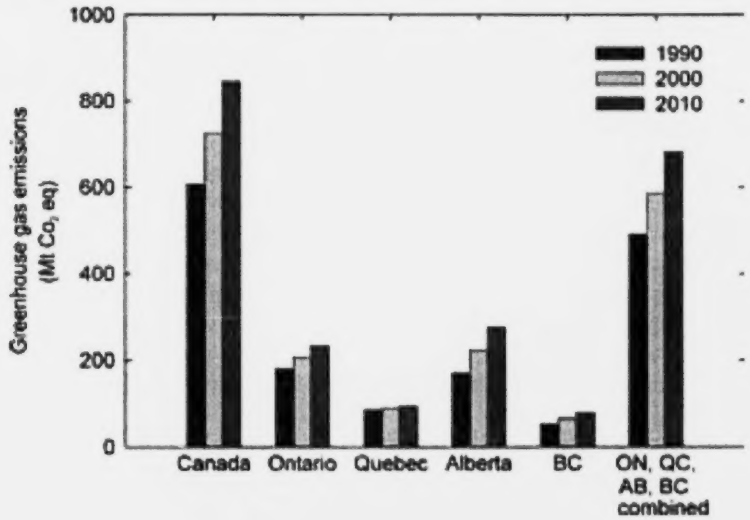

Figure 1.1. Greenhouse gas emissions in CO_2 equivalents (CO_2eq) for Canada and individual provinces: Ontario (ON), Quebec (QC), Alberta (AB), and British Columbia (BC) in 1990, 2000 and 2010. Emissions projected for 2010 are based on the rate of growth in emissions from 1990 to 2000. (Data obtained from Environment Canada web site www.ec.gc.ca/pdb/ghg/1990_00_report/ accessed November 17, 2003.)

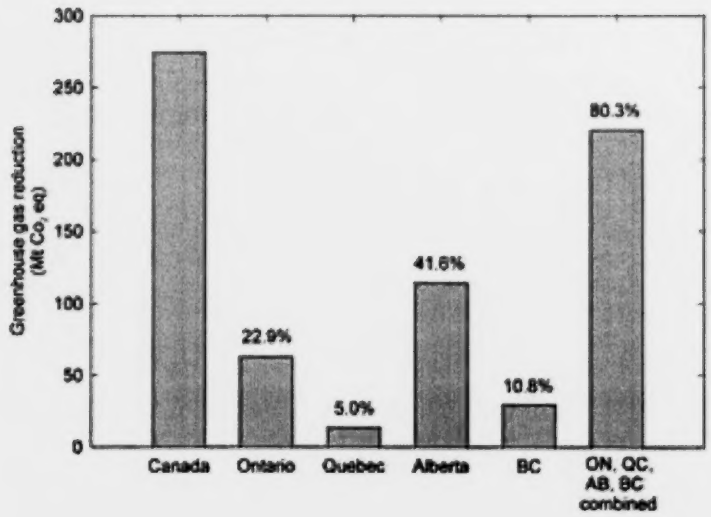

Figure 1.2.
Projected reductions in C emissions required in 2010 to achieve a reduction in emissions of 6% below the 1990 level, based on projected 2010 emissions presented in Fig. 1.1. Numbers above vertical bars are the percent of the total Canadian emission reductions potentially required in Ontario (ON), Quebec (QC), Alberta (AB), and British Columbia (BC) in 2010. (Derived using data obtained from Environment Canada web site www.ec.gc.ca/pdb/ghg/1990_00_report/ accessed November 17, 2003.)

While Canada will need to meet most of its commitment through emission reduction strategies, some land use activities such as forest management, afforestation, and agricultural soil management can be used to help meet the Kyoto Protocol target. Article 3.3 of the Kyoto Protocol requires its signatories to report on the change in C storage resulting from the net land use change due to afforestation and deforestation. If the net balance is positive (i.e., a C sink exists) this may be used to offset C emissions from other man-made sources of GHGs. However, if the net land use balance is negative (i.e., deforestation has created a net release of GHGs to the atmosphere), then the emission must be accounted for in the total emission balance. Article 3.4 of the Kyoto Protocol is optional and allows a finite amount of C storage in forests resulting from forest management to offset a release due to deforestation or industrial sources. This report addresses practices that can increase forest C storage under Article 3.4. If Canada implements Article 3.4 for its forests, it would require accounting for the total net change in C. Thus, reduced forest C from natural disturbances such as fire and insects could potentially make forests a source of C, even if the effects of forest management otherwise increased C storage. Clearly, this reduces the attractiveness of implementing Article 3.4. A national risk analysis is being prepared by the Canadian Forest Service and will be used to inform Canada's decision on this component of the Kyoto Protocol.

Viewed from a biogeochemical perspective and in the context of climate change and the Kyoto protocol, the primary focus of forest management is manipulation of the forest C cycle to increase the rate of C accumulation in usable forms of biomass (i.e, stems, branches, etc.) without sacrificing long-term productivity. Various silvicultural practices are applied within the forest management cycle to increase photosynthetic CO_2 uptake (and therefore growth and biomass accumulation) by modifying the growth environment and improving resource availability (Fig. 1.3). For example, control of competing vegetation improves stand growth, establishment, and biomass C storage by increasing light, water, and nutrients available to crop trees (Stewart et al. 1984, Walstad and Kuch 1987). Alternatively, increasing growth and C storage of some forests through more intensive management acts to maintain the C sequestered in other forests that are best preserved or protected for values other than wood production. Forests also contribute indirectly to the reduction in CO_2 emissions to the atmosphere by producing wood that (1) replaces energy-intensive (and therefore CO_2 emitting) structural materials such as steel and (2) replaces fossil sources of fuel (Nabuurs 1996, Marland and Schlamadinger 1999). As a consequence, forest management may reduce the amount of CO_2 in the atmosphere

by augmenting C stored in trees, forest soils, and wood products (Sedjo 1989, Schroeder 1991, Binkley et al. 1997). Carbon stored in wood products will not be part of the first Kyoto forest C accounting period.

Because of the large forest area in Ontario, even small changes in C per hectare caused by forest management can either sequester or release large amounts of C. The objective of this report is to quantify the potential consequences of a suite of forest management practices on the C balance of forests in Ontario. This report is organized based on the four-part forest management cycle that includes stand establishment, growth enhancement, forest protection, and harvesting (Fig. 1.3). The report summarizes information from the scientific literature for a variety of specific intensive forest management activities. These practices are first defined relative to their operational use in Canada, followed by a review and synthesis of their effects on tree and stand growth. This discussion is focused on commercially important tree species of northern forest ecosystems where possible. The effect of these practices on C sequestration in biomass, dead organic matter (DOM), and soils is then examined using information reported in the scientific literature and/or projected using model simulations. In the first part, (Section 2) the stand establishment practices of site preparation, planting, and vegetation management are addressed. Section 3 focuses on forest management practices that enhance stand growth, and includes genetic improvement, thinning, and fertilization. The implications of protection from forest fire, insects, and disease on forest C storage are described in Section 4, while Section 5 assesses the impacts of forest harvesting on C storage, and includes the construction of roads, landings, and skid trails, as well the influence of rotation length. A synthesis of these management practices on forest C storage to estimate the potential overall gains in C storage that might be attained through their use in Ontario is provided in the final section (Section 6).

Projections of the effects of selected forest management practices on C storage are made using generalized soil and litter, or DOM, C values for eastern Canadian boreal conifer sites reported by Kurz et al. (1992, 1996) and Apps et al. (1993, 1999, 2000). These soil and DOM values are combined with growth and yield data from Plonski (1971) to produce stand-level C estimates. This numerical approach is designed to illustrate where forest management is most likely to result in large changes in forest C stocks. More accurate predictions of how forest management affects C require that forest species composition and site class is modelled at finer geographic scales. This work is underway in a separate project to quantify C budgets of Ontario's forests at the forest management unit, regional, and provincial levels (Boivin et al. 2005).

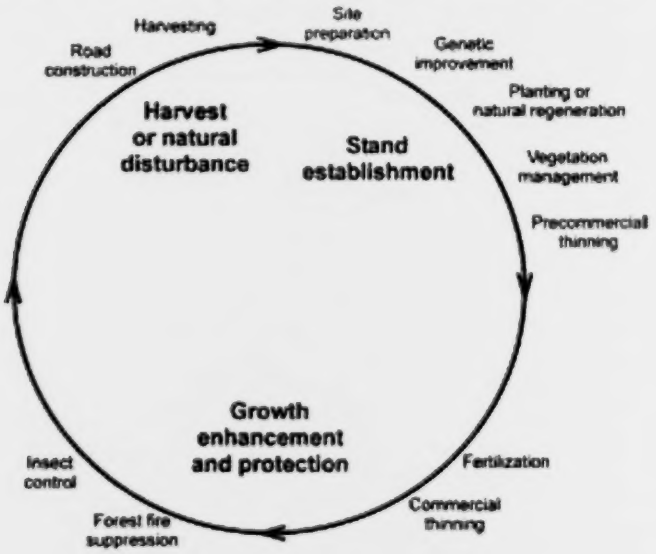

Figure 1.3.
The forest management cycle. A stand-replacing disturbance (either natural due to fire, insects, or severe weather or human-caused due to harvesting) and stand establishment practices occur sequentially within several years of one another. Growth enhancement and protection practices can occur over most of the forest management cycle. (Modified from Lautenschlager 2000).

Ontario's Forest Resources

Ontario comprises more than one million square kilometres, of which about 65% (70.5 million ha) is forested (OMNR 2002a). About 82% (57.8 million ha) is under Crown ownership, 8.5% (6.0 million ha) in parks and protected areas, and about 10% (6.8 million ha) is federally or privately owned. About 80% of this forested land (55.7 million ha, or roughly half of all the land area in the province) is considered productive for the growth of timber (OMNR 2002a). This area represents 1.3% of the world's productive forest and is approximately twice the total area of Great Britain.

Four major forest types (called Forest Regions) occur in Ontario: the Hudson Bay Lowlands, the Boreal Forest, the Great Lakes–St. Lawrence (GLSL) Forest, and the Deciduous Forest (Fig. 1.4, Table 1.1) (OMNR 2002a). Approximately 55% of Ontario's forests consist of relatively pure conifer stands, 20% are pure broadleaved, deciduous stands, and 25% are mixedwoods. Black spruce (*Picea mariana* (Mill.) BSP) is the most common tree species in Ontario's forests (41% of the growing stock by gross total volume[3]), followed by poplar (trembling aspen, *Populus tremuloides* Michx.; large-toothed aspen, *P. grandidentata* Michx.; balsam poplar; *P. balsamifera* L.) (17%), jack pine (*Pinus banksiana* Lamb.) (13%), sugar maple (*Acer saccharum* Marsh.) (8%), white birch (*Betula papyrifera* Marsh.) (7%), and balsam fir (*Abies balsamea* L.) (4%) (OMNR 1996, 2002a).

[3] The Ontario Ministry of Natural Resources and defective and decayed wood.

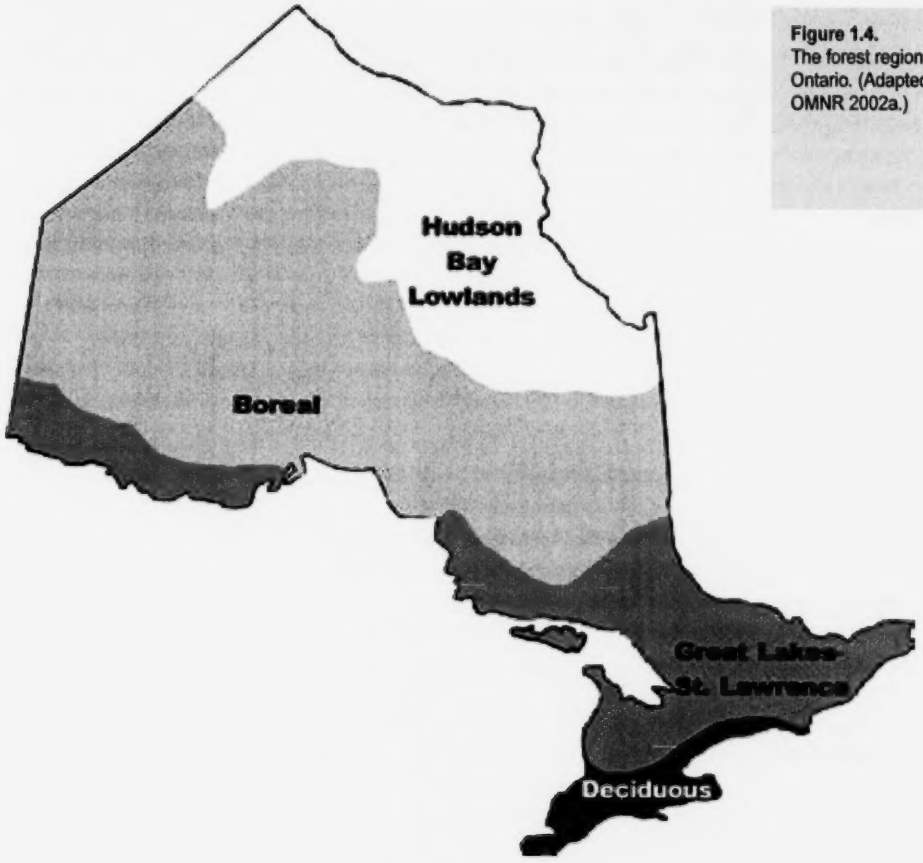

Figure 1.4.
The forest regions of Ontario. (Adapted from OMNR 2002a.)

Table 1.1. Total forest, productive forest, productive crown forest, and total land area (thousand ha) in the major forest regions of Ontario (OMNR 2002a)*. Numbers in parentheses are proportions of total area (%) within a given classification.

Region	Forest	Productive forest	Productive Crown forest	Total area
Hudson Bay Lowlands	14,509 (21)	4,818 (9)	4,505 (10)	25,629 (26)
Boreal Forest Region	41,653 (59)	37,165 (67)	32,488 (73)	49,886 (51)
GLSL Forest Region	13,843 (20)	13,301 (24)	7,321 (17)	20,142 (20)
Deciduous Forest Region	486 (<1)	452 (<1)	87 (<1)	3,048 (3)
Total	**70,491**	**55,736**	**44,401**	**98,705**

*Does not include area covered by the Great Lakes. Total land area includes forested and non-forested land.

The Hudson Bay Lowlands, located in the northernmost part of Ontario, cover about 26 million ha, and consist mostly of poorly drained muskeg, with trees usually occurring in scattered patches of stunted black spruce and tamarack (*Larix laricina* (Du Roi) K. Koch). The soils are usually poorly drained and permafrost is widespread. Due to limited site productivity and sub-arctic climate, forests in this region are not managed except to protect communities and infrastructure from fire.

The Boreal Forest Region covers more than half of Ontario, about 50 million ha, stretching in a continuous 1,100 km band from Manitoba in the west to Quebec in the east, averaging more than 500 km from north to south. Short summers and cold winters characterize this forest region, with a mean annual temperature of approximately 0°C. Average summer temperature is 14°C and mean winter temperature is –12°C. Annual precipitation ranges from 400 mm in the northwest to 900 mm in the southeast. Wetlands and poorly drained forested lands cover nearly 25% of the region, with their concentration increasing towards the north (OMNR 2002a). Pure stands of jack pine or mixtures of jack pine, trembling aspen, and paper birch commonly inhabit drier sites. Black spruce mixed with lesser amounts of balsam fir, tamarack, and eastern white cedar (*Thuja occidentalis* L.) dominate poorly drained areas. Forest fire, insects, and harvesting shape the structure and function of the boreal forest and often result in pure, even-aged stands. The dominant silvicultural system in the boreal forest is clearcutting. Planting or seeding are used where natural regeneration is not expected to provide well-stocked stands of commercial tree species. Of the average of 206,000 ha harvested annually in Ontario from 1990 to 2003 most were in the boreal forest and more than 90% was clearcut (CCFM 2005). Gross total volume of growing stock in this region was estimated to be 3,939 million m^3 in 1995 (OMNR 1996).

The Great Lakes-St. Lawrence Forest Region covers about 20 million ha. Forests in this region are mainly located in south-central Ontario but also extend into northwestern Ontario and along the northern shore of Lake Huron and Lake Superior. Warm summers and mild winters characterize this region, where mean annual temperature is 6°C, average summer temperature is 17°C, and mean winter temperature is –4.5°C. Annual precipitation ranges from 750 to 1000 mm. Tree species in the region include sugar maple (27% of gross total volume), poplar (18%), eastern white pine (*Pinus strobus* L.) (8%), white birch (8%), and jack pine (7%), with yellow birch (*Betula alleghaniensis* Britt.), black cherry (*Prunus serotina* Ehrh.), red maple (*Acer rubrum* L.), red oak (*Quercus rubra* L.), basswood (*Tilia americana* L.), white ash (*Fraxinus americana* L.), American beech (*Fagus grandifolia* Ehrh.), and eastern hemlock (*Tsuga canadensis* (L.) Carr.) occurring less frequently (OMNR 1996). The selection and shelterwood silviculture systems are the main methods used in the management of GLSL forest ecosystems. The total growing stock in this region in 1995 was 1,400 million m^3 (OMNR 1996).

Prior to European settlement, the Deciduous Forest Region covered over 3 million ha. Over the past 150 years, this forest has been displaced by agriculture and urbanization in southwestern Ontario. Less than 16% of the original area is now forested. Remnants of this forest region survive mainly as small pockets on private land of low agricultural potential (e.g., sand plains, bottomlands) and within provincial parks along the north shore of Lake Erie. The climate of the region is characterized by humid, warm summers and mild winters. The mean annual temperature is approximately 8°C. Kentucky coffee tree (*Gymnocladus dioicus* (L.) K. Koch), sassafras (*Sassafras albidum* (Nutt.) Ness), flowering dogwood (*Cornus florida* L.), and several species of oak (*Quercus sp.*) and hickory (*Carya sp.*) are among the most common Carolinian tree species in this forest region.

Most of Ontario is subject to some form of forest management (Fig. 1.5). The Planning Area covers some 45.3 million ha, 83% of which is forested, and is the context area for Ontario's Living Legacy (OMNR 1999). This zone lies within the area covered by Ontario's Forest Resource Inventory, and is the area where formal forest management planning is currently practiced in 47 forest management units (FMU). The Northern Boreal Zone encompasses the Hudson Bay Lowlands and a significant portion of the Boreal Forest Region, most of which lies north of 51° N latitude. Within this region, forests are influenced primarily by natural disturbances, with very little management being applied other than for protection of remote communities and infrastructure from wildfire. The Southern Ontario Zone includes small isolated fragments of the GLSL Forest Region and all of the Deciduous Forest Region.

Figure 1.5.
Managed forest in Ontario. Areas included in the enclosed polygons are Crown forest management units, parks, or inventoried forests in the far north that may become future sites of forest operations (obtained from Mike St. Eloi, Ministry of Natural Resources).

Why Practice More Intensive Forest Management?

Forest management follows a cyclical pattern of forest development that begins with stand renewal following natural disturbance by forest fire, insects, and disease, and the human disturbances caused by timber harvesting. Increases in the intensity of forest management usually reflect decisions to obtain more resources from the forest. Forestry in Ontario is carried out to support a variety of interests, including the supply of timber. However, forest management intensity is most often increased with the intent of sustaining the supply of conifer tree species to forest industry, and this objective has in many ways shaped the practice of forestry in Ontario (Armson et al. 2001). For example, increases in forest management efforts resulted in the expansion of bareroot tree nurseries in the 1950s and container nurseries in the 1980s. In addition, it lead to the creation of Ontario's provincial research branch and produced many technical innovations related to reforestation, including the Brohm aerial seeder and the Ontario Tube (one of the world's first container tree seedling systems) (Armson et al. 2001).

Forest management has recently been subdivided into four basic categories with respect to their current and potential future use in Ontario: extensive, basic, intensive, and elite (Bell et al. 2000). The levels of forest management represent a gradient in management intensity that is devoted to affecting the species composition and growth rate of the forest. *Extensive forest management* provides protection from fire and insects and relies largely on unassisted natural regeneration, with little concern given to future forest composition. *Basic forest management* is similar to extensive management, but in addition to forest protection, regeneration can be promoted through planting, seeding, and assisted natural regeneration where needed, as well as site preparation and vegetation management. This level of management seeks to control species composition and increase crop tree growth rate of the future stand. *Intensive forest management* combines basic forest management with stand density regulation at a young age (i.e., juvenile spacing, pre-commercial thinning) to further increase growth rate of desired crop species. *Elite forest management* combines intensive management with several practices aimed primarily at increasing the value and/or volume of desired forest components (not just timber), and in addition to the previously mentioned practices can include commercial thinning, pruning, genetic improvement, fertilization, and drainage/irrigation. In the cases of fertilization, drainage, and/or irrigation, management efforts seek to increase the inherent site productivity of the forest.

Using these definitions, it has been estimated that forests in the Planning Area are receiving extensive, basic, and intensive management over 32%, 30%, and 2% of the total area, respectively, with the remainder allowed to respond to natural disturbance with little human intervention (Wayne Bell, 2005, OMNR, pers. comm.). In the Southern Ontario zone, where most of Ontario's population resides and much of the original forest area has been displaced by agriculture and urbanization, forest management is being applied as 30% extensive, 35% basic, 12% intensive, and 3% elite, with 20% unmanaged. In the largely uninhabited Northern Boreal Forest zone, about 1% of the forest area receives extensive management where remote communities harvest fuel wood.

Continuing pressure on the forest resource to provide wood fibre along with many other services (e.g., recreation, wildlife habitat) to the public has resulted in the need to increase wood production on an ever-diminishing forest land base (Nambiar 1996, Wagner et al. 2004). Practicing more intensive forest management is one obvious approach to maintaining or increasing future wood supply in the face of continued reduction in forest area open to timber harvest. This echoes suggestions by South (1996) and Binkley (1997) who recommend that the best way to preserve and protect natural forests from harvesting is to intensify management on select, highly productive areas. Recently, 2.4 million ha of new parks and protected areas were created under Ontario's Living Legacy (OMNR 1999), an historic agreement between the Crown, forest industry, public, and other user groups. This agreement affected about 6% of the Planning Area, removing about 5% of the potential provincial wood supply (OMNR 1999). The Ontario Forest Accord, which stated that no net loss of fibre was to result from this land removal, was part of this agreement. More intensive forest management approaches to maintain this fibre yield and the designation of enhanced forest productivity areas are being considered (OMNR 1999). Associated changes may have implications for forest C storage.

Managing the Forest Carbon Cycle to Increase Sequestration

The carbon cycle of natural forests

Globally, forests store tremendous amounts of C in four major pools: living biomass, coarse woody debris (CWD), organic soil horizons, and mineral soil. Tropical, temperate, and boreal forests cover 4.2 billion ha and biomass and soils contain 359 and 787 Gt of C, respectively (Dixon et al. 1994). For boreal forests, the average C density (t ha^{-1}) is 407 t ha^{-1}, with 64 t ha^{-1} (16% of total) in biomass and 343 t ha^{-1} (84% of total) in soils (Dixon et al. 1994). In unmanaged forests, atmospheric C in the form of CO_2 enters this forest C pool through plant photosynthesis, and eventually moves to living and dead biomass C pools, the soil C pool, or back to the atmosphere through several processes. At the stand or ecosystem level, the net C balance of the forest is dependent on the processes controlling net primary productivity (NPP) and net ecosystem productivity (NEP). Net primary productivity refers to the rate at which C is incorporated into living biomass and is equal to gross primary productivity (GPP) (or gross photosynthesis) minus C losses to autotrophic respiration (Ra) of living plant tissues. The rate of total ecosystem C accrual, NEP, can be positive or negative, and is equal to GPP minus Ra and heterotrophic respiration (Rh), the C losses associated with the decomposition of dead biomass. The amount of C in each pool and the rates of C flux between C pools estimated for a 115-year-old black spruce forest ecosystem are presented in Figure 1.6. In this example, GPP exceeds total respiratory C losses (Rt, the sum of Ra and Rh), NEP is positive (0.68 t C ha^{-1} yr^{-1}), and the forest ecosystem is acting as a C sink.

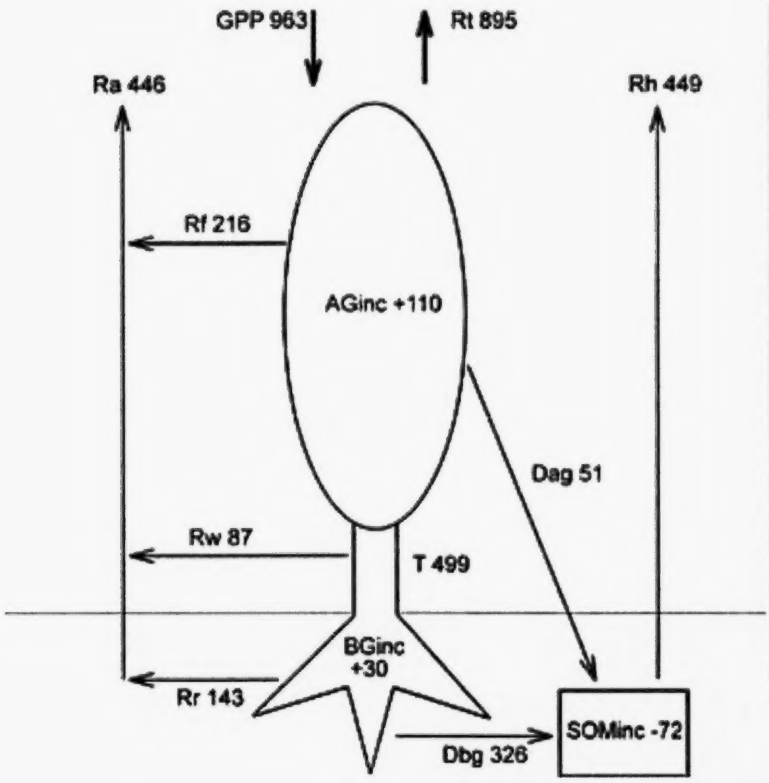

Figure 1.6.
Estimated total annual carbon fluxes for a 115-year-old boreal black spruce forest ecosystem located in Prince Albert, Saskatchewan. Values for above- and belowground C fluxes are from Gower et al. (1997) and Steele et al. (1997), respectively. All units are g C m^{-2} yr^{-1}. Abbreviations: Gp, gross photosynthesis; Rt, total respiration; Ra, autotrophic respiration; Rh, heterotrophic respiration; Rf, foliar respiration; Rw, aboveground wood respiration; Rr, root respiration; Dag, dead aboveground organic matter (litter and mortality); Dbg, belowground dead organic matter (root mortality, fine root turnover, and exudation); T, belowground translocation; AGinc, aboveground net biomass increment; BGinc, below-ground net biomass increment; SOMinc, net soil organic carbon increment. (Adapted from Malhi et al. 1999.)

Rates of the primary processes controlling the C balance and forest biomass development vary considerably with stand age, or time after a stand-replacing disturbance (Fig. 1.7). Initially, forests act as C sources (NEP is negative) when C losses to DOM decay (Rh) are larger than C gained through NPP (Fig. 1.7a). Fluxes of C from DOM decay are primarily from decomposition of forest floor detritus and dead root tissues since mineral soil C levels remain relatively constant following disturbance (Johnson and Curtis 2001). Stand NPP increases with stand leaf area until canopy closure occurs, which contributes to the concomitant reduction in DOM decay. When NPP exceeds total respiration losses (Rt), the stand becomes a net sink for C (NEP is positive). During this early stage of stand development, much of the variability in NEP is related to factors controlling DOM decay, such as litter chemical composition and soil temperature, moisture content, and oxygen levels. In comparison, NPP is dependent on the dominant vegetation type, and site and climatic factors limiting photosynthesis and GPP (i.e., light, temperature, and soil moisture). NPP is highest in young stands, increases to a peak value at intermediate stand age, and declines thereafter (Ryan et al. 1997, Pregitzer and Euskirchen 2004). Generally, variation in NEP roughly parallels that of NPP but is more strongly related to the frequency and intensity of disturbance than to climatic factors (Chapin et al. 2002, Pregitzer and Euskirchen 2004). Therefore, natural disturbance regimes influence the forest C balance and efforts to increase forest C storage, primarily by affecting the age at which stand NEP becomes positive.

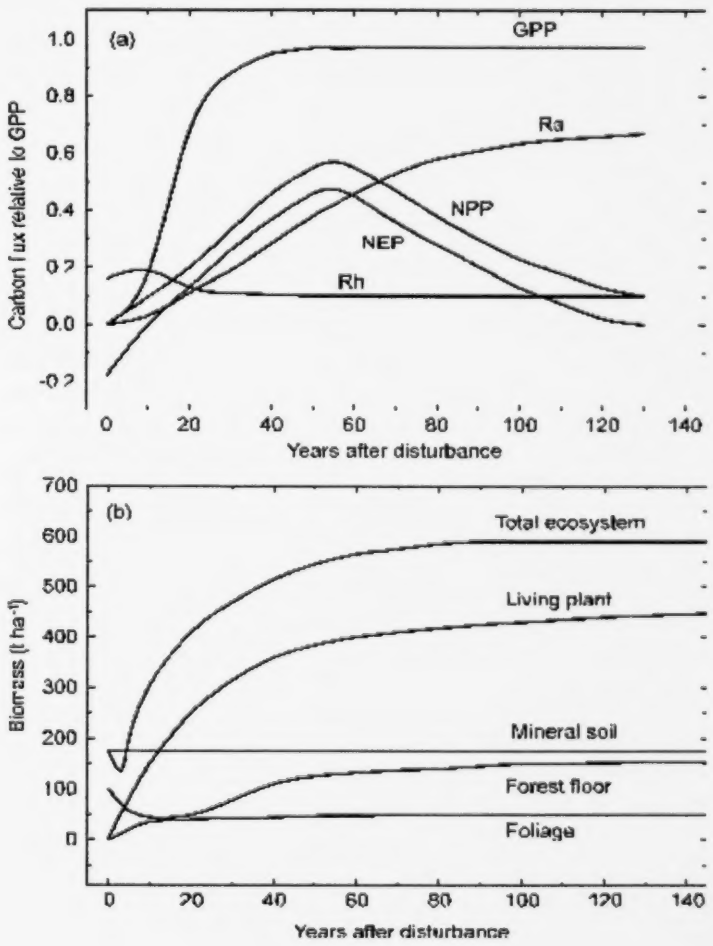

Figure 1.7.
Generalized relationship of (a) major forest fluxes (NPP, NEP, Ra, Rh) relative to that of GPP and (b) live or dead biomass accumulation in foliage, forest floor, mineral soil, living plants, and total ecosystem following a stand-replacing natural disturbance (e.g., wildfire). Abbreviations for (a) are defined in Figure 1.6 caption. (Adapted from Barnes et al. 1998.)

The general pattern of biomass accumulation with stand age is illustrated in Figure 1.7b. Biomass accumulation in living plant tissues begins almost immediately after disturbance as plants begin to colonize the site via a variety of sexual and vegetative reproductive strategies. Coincident with this increase in plant biomass is an increase in foliar biomass, which attains a steady-state value as canopy closure of the developing vegetation occurs. Dead biomass in the mineral soil changes very little after disturbance, while the mass of forest floor organic layers declines for a period after disturbance when DOM decay is near its maximum and biomass losses to decomposition are larger than inputs from litter fall. Organic layers begin to increase when litter inputs exceed the decomposition of this material. Total ecosystem and living biomass reach their maximum value and assume a steady state at an intermediate stand age where peak NPP and NEP also occur. Although stand NPP and NEP decline with age, total ecosystem biomass per unit area remains relatively stable due mostly to the accumulation of CWD (Hély et al. 2000). As much as 75% of the forest C of older stands may be present as dead biomass in the forest floor and mineral soil (Kasischke et al. 1995, Kurz and Apps 1999, Kasischke 2000). The amount of biomass contained in living plant tissue may fluctuate slightly over time as mortality of even-aged dominant individuals in the overstory occurs. During this transitional period of stand development, the release and canopy recruitment of advance regeneration of later successional species occupying canopy gaps begins to move the stand towards an all-aged stand condition (Oliver and Larson 1996).

The carbon cycle of managed forests

The primary difference between the C cycles of natural and managed forests is the harvesting, processing, and transfer of forest C into wood products. The activities associated with harvesting and the subsequent regeneration of the stand also influence the NPP, NEP, and C storage of managed forest ecosystems dependent on the frequency and intensity of these human-caused disturbances. The goal of forest C sequestration efforts is to reduce the amount of forest C entering the atmospheric pool using four basic mitigation mechanisms: (1) increasing the amount of C in forest ecosystems, (2) increasing the period of C storage in forest products, (3) substituting forest biomass for fossil fuels (i.e., direct substitution), and (4) substituting forest products for more energy-intensive structural materials such as aluminum, steel, concrete, and plastics (i.e., indirect substitution) (Apps et al. 1999). Although the forest products sector holds a small portion of the total forest sector C (<5%), it can contribute significantly to mitigation efforts (Apps et al. 1999) (Fig. 1.8). Thus, a complete accounting of forest management effects on C cycling within the forest sector must include a forest products component (Apps et al. 1999, Thornley and Cannell 2000, Liski et al. 2001, Harmon and Marks 2002).

In managed forests, C may be removed from the stand by harvesting or modified by management activities and disturbances that alter the relative transfer of C to biomass and soil. Forest harvesting results in tree biomass C being transported to mills for processing into wood products, with a small proportion left on site as logging slash (Fig. 1.8). This forest C remains in the wood products pool as long as the product is used, and then depends on its fate and decay rate following disposal. Most paper products are relatively short-lived (<6 years), after which they are recycled, burned, or disposed of in landfills. Solid wood products such as lumber and plywood are stored for comparatively long periods (30 to100 years) in furniture, buildings, and other structural materials, and are eventually burned or deposited in landfills. Wood residues produced during processing are often burned for energy, the amount varying with the type of product and the efficiency with which the raw material is used.

Forest carbon in wood products

Winjum et al. (1998) estimate that 9 Mt of wood products were manufactured in Canada in 1990, about 75% of which had a service lifetime greater than five years. In Ontario in 1995, timber harvest was about 26.3 million m^3 (CCFM 1997). Of this total, 16.3 million m^3 (62%) was used in solid wood products (e.g., lumber,

plywood, and veneer), 8.4 million m^3 (32%) was used for paper products, and 1.6 million m^3 (6%) was used for firewood (CCFM 1997). In contrast, the timber harvest in Quebec in 1995 was 41.7 million m^3, which was divided into 33.6 million m^3 (81%) of solid wood products, 5.9 million m^3 (14%) of pulpwood, and 2.2 million m^3 (5%) of firewood (CCFM 1997). In British Columbia in the same year, the total harvest was 74.5 million m^3, of which more than 99% was processed into solid wood products (CCFM 1997). Altering forest management practices (e.g., lengthening rotations, thinning, fertilization) to produce larger trees will provide wood that is more valuable for use in solid wood products than paper and result in longer storage periods for forest C. A complete forest C budget model for Ontario should include the forest products C pools, making it possible to predict how changes in wood use could contribute to reducing C emissions.

Forest management, wood properties, log quality, and carbon storage in forest products

Forest management is usually conducted to increase the quantity and/or quality of wood produced per unit land area (Brazier 1977, Bendston 1978). However, some management practices can adversely affect the strength and physical attributes of stem wood, its technical suitability for lumber or pulp, and therefore the duration of C storage in forest products. Generally, managed forests are characterized by accelerated growth, delayed canopy closure, and shortened rotation length. As a result, trees from managed forests tend to contain a greater proportion of juvenile wood, with a lower specific gravity, shorter tracheids, and more compression wood. All of these are undesirable features for solid wood products and lower the strength and pulp yield per unit wood volume (Fielding 1967, Larson 1968, Brazier 1977, Bendston 1978). Planting at wider spacing and excessive thinning can increase crown width, stimulate epicormic branching, and/or delay natural pruning of lower branches. These tree responses can reduce the quality of wood for use as lumber or pulp by forming stems with more taper, shorter clear, knot-free lengths, and larger knot size (Fielding 1967, Larson 1968, Janas and Brand 1988, Heitzman and Nyland 1991, Barbour et al. 1994). The presence of these defects can be minimized by pruning or more frequent, less intensive thinning. Despite the influence of forest management on wood properties, the increase in wood volume provided by intensive forest management far outweighs any detrimental effects of reduced wood quality (Larson 1968, Brazier 1977, Bendston 1978).

Forest carbon in landfills

The length of time C is stored in wood products during their use has a relatively minor effect on C sequestration as a climate change mitigation option (Skog and Nicholson 1998, Harmon and Marks 2002). It is instead the relative proportions of wood products that are burned, recycled, or enter landfills at the end of their life that affects the forest and wood products C cycle (Micales and Skog 1997, Apps et al. 1999). Burning returns biomass C to the atmosphere and, where used as a substitute for fossil fuels, can significantly reduce C emissions. Recycling can reduce the amount of fossil fuel energy consumed and emissions produced relative to that required to manufacture new material. For example, production of newsprint from recycled rather than virgin material reduces energy consumption and emissions by 40 to 45% (Mohareb et al. 2004). Recycling of forest products also decreases the area of forest that needs to be harvested, thereby maintaining the area of active forest C sinks on the landscape.

Modern landfills are becoming an increasingly important sink for forest C that, unlike forest ecosystems, is unaffected by any projected changes in climate (Apps et al. 1999). Landfills store forest C almost indefinitely as a result of the slow decay rate of wood products in this anaerobic environment (Micales and Skog 1997, Skog and Nicholson 1998). Lignin is not metabolized by anaerobic bacteria and therefore does not decompose significantly in landfills (Micales and Skog 1997, Mohareb et al. 2004). For highly lignified products, such as solid wood and newsprint, as much as 97% and 84% of C held in these products, respectively, are sequestered in landfills indefinitely. For paper products with lower lignin content and larger amounts of more readily

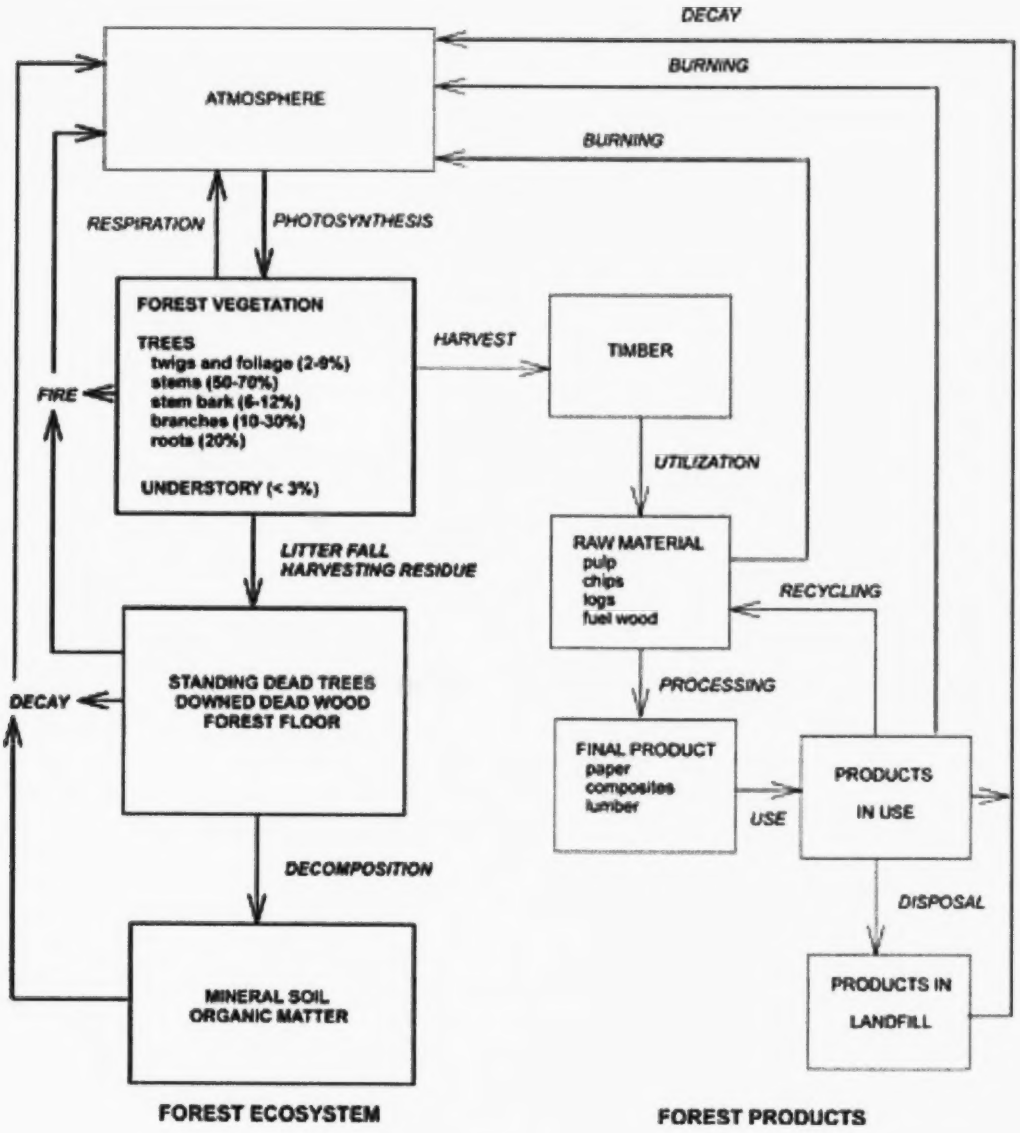

Figure 1.8.
The forest sector C cycle showing the forest ecosystem and forest products components (modified from Parker et al. 2000). Carbon fluxes (arrows) and pools (boxes) with heavy lines are modelled in the simulations carried out for this report.

decomposed cellulose and hemicellulose, up to 60% of their C will remain permanently sequestered. In general, less than 50% of the C present in wood or paper in landfills is ever released as CO_2 or CH_4. Of this C released, about 40% is emitted as CO_2 and 60% as CH_4 (Skog and Nicholson 1998, Mohareb et al. 2004). These gases are released gradually over a 5- to 40-year period. Although CH_4 has a greater 'greenhouse gas effect' than CO_2, CH_4 generated by decomposition of wood in landfills can be captured and burned for energy as it is released, at which time it is converted to CO_2 (Mohareb et al. 2004). The C that is not converted to landfill gases is eventually transformed into an inert solid mass similar in chemical composition to humic materials (Micales and Skog 1997).

In general, greater C storage in wood products can be achieved by increasing product life, recycling, and increasing the proportion of more lignified materials that are produced (Skog and Nicholson 1998). The benefits of C storage in forest products are determined by the ratio of the decay rate of these products relative to the decay rate of an equal mass of dead biomass (Hendrickson 1990). However, the largest mitigation effect is achieved when biomass C is used in either direct or indirect substitution (Kohlmaier et al. 1998, Richardson et al. 2002). For example, the use of residual biomass for energy production can significantly reduce fossil fuel consumption and associated CO_2 emissions (Zerbe 1993, Klass 1998. Richardson et al. 2002). Alternatively, the use of wood in place of more energy intensive materials can reduce the relatively large CO_2 emissions generated during their manufacture (Koch 1992, Kohlmaier et al. 1998). For example, construction of concrete-framed and steel-framed residential homes consumes 15 to 17% more energy than wood-framed homes, and the global warming potential of emissions associated with these buildings are 26 to 31% higher for concrete- or steel-framed homes (Lippke et al. 2004).

Ontario's Forest Carbon Budget

The historic C budget of Ontario's forests was estimated by Peng et al. (2000) using the Carbon Budget Model-Canadian Forest Sector, version 2 (CBM-CFS2) developed by Kurz and Apps (1999). This model simulates the dynamics of groups of forest stands that are similar in species composition, productivity, stocking, and age-class structure using growth curves from forest inventory data. Allometric formulae are used to predict above- and belowground biomass from merchantable stem volumes. Soil and DOM C in CBM-CFS2 are modelled using equations that estimate additions to forest floor litter, releases from disturbances, and their decomposition based on forest type (e.g., hardwood and conifer), climate, and stand age. The biomass C in Ontario's forests predicted by Peng et al. (2000) is based on national forest inventory data (Bonnor 1985) that encompass the entire forested area of Ontario, including areas outside of the Crown FMUs (see Figure 1.5).

According to Peng et al. (2000), Ontario's forests contained approximately 12,648 Mt of C in 1990 (Table 1.2). Of this, 13% (1,702 Mt) of the forest C was stored as biomass and 87% (10,946 Mt) was stored in the soil and DOM (Table 1.2). Averaged for all Ontario forests in 1990, the C density was 179 t C ha^{-1} (Peng et al. 2000) with an average of 24 t ha^{-1} in biomass, and 155 t C ha^{-1} in soil and DOM. About 88% of Ontario's forest C was estimated to reside in the Boreal Forest Region, 11% in the GLSL Forest Region, and 1% in the Deciduous Forest Region (Peng et. al. 2000). The low total C in the Deciduous Forest Region is due primarily to its limited area and belies its sequestration potential: The average C density of the Deciduous Forest Region is estimated to be 336 t ha^{-1}, compared to 191 and 177 t ha^{-1} in the GLSL and Boreal Forest Regions, respectively (Peng et al. 2000).

Table 1.2. Carbon storage in Ontario forests in 1990 (from Peng et al. 2000). Fast, medium, and slow refers to the decay rate of different pools of soil and DOM C as defined by Kurz and Apps (1999). Row and column totals may not equal the sum of the row values due to rounding.

Forest region	Total C (Mt)	Biomass C (Mt)			Soil and DOM C (Mt)			
		Above-ground	Below-ground	Total	Fast pool	Medium pool	Slow pool	Total
Boreal	11,097	1,037	299	1,336	340	1,841	7,580	9,761
GLSL	1,484	264	71	336	34	242	872	1,148
Deciduous	67	25	5	30	1	10	26	37
Total	12,648	1,326	376	1,702	376	2,092	8,478	10,946

Estimating the Effects of Management on Carbon in Ontario Forests

In this report we provide estimates of the potential major effects of forest management on the amount of C stored in Ontario forests. For some forest management practices, we relied on information available in the scientific literature to make qualitative judgments about the influence of these activities on forest C storage. For other management practices, we used a more quantitative approach. For the latter, a generalized forest soil, litter, and biomass scenario was developed using stand yield data from Plonski (1971) for Site Class 2 black spruce in Ontario. Stand yield data were input to CBM-CFS2, currently being used to model the C budgets of forests in Ontario and elsewhere in Canada (Kurz et al. 1992, Kurz and Apps 1999, Peng at al. 2000). This model permits users to assess the relative magnitude of changes in forest C that result from forest management. The components of this model are described below by way of an example comparing the effects of clearcut harvesting and wildfire on forest C cycling. Modelling to produce more precise estimates is warranted but beyond the scope of this report.

Life-cycle assessment or analysis is a tool used to quantify the influence of a product on the environment, from extraction of raw materials, conversion, manufacture, transport and use, to its disposal including waste management (Kohlmaier et al. 1998, Brentrup et al. 2001, Berg and Karjalainen 2003, Lippke et al. 2004). This approach has been used to estimate total GHG and CO_2 equivalent emissions associated with fossil fuel use during various forestry operations in Scandanavia (Karjalainen and Asikainen 1996, Berg 1997, Berg and Karjalainen 2003). For example, in Finland most emissions (57%) associated with forestry operations are from permanent forest road construction and long-distance transportation of fibre to processing centres, with much less emitted by felling (13%), skidding (18%), and silvicultural activities such as scarification and tending (<8%) (Karjalainen and Asikainen 1996). These emissions amount to roughly 2.5 to 3.0 kg C per m^3 of wood harvested, or about 1.4% of C contained in harvested tree biomass (Berg and Karjalainen 2003). Emissions associated with forestry activities also increase with the level of mechanization involved and the frequency of silvicultural intervention (Karjalainen and Asikainen 1996, Berg 1997, Liski et al. 2001). Emissions also vary with the manufacturing process, and are greater for pulpwood than solid wood products (Liski et al. 2001). This type of analysis was beyond the scope of the present project, but emissions associated with a few forestry activities are briefly addressed in this report.

Soil and dead organic matter carbon

Forest soils (defined here as the mineral soil, non-decomposed dead belowground organic matter, and humus) and DOM (dead aboveground biomass, including litter, and fine and coarse woody debris) together usually contain more C than forest biomass. Therefore, practices affecting soil conditions and the accumulation or decomposition of DOM are likely to affect C whether forest management causes the net movement of C into or out of the forest. Soil and DOM C in CBM-CFS2 are estimated from empirically derived functions for biomass

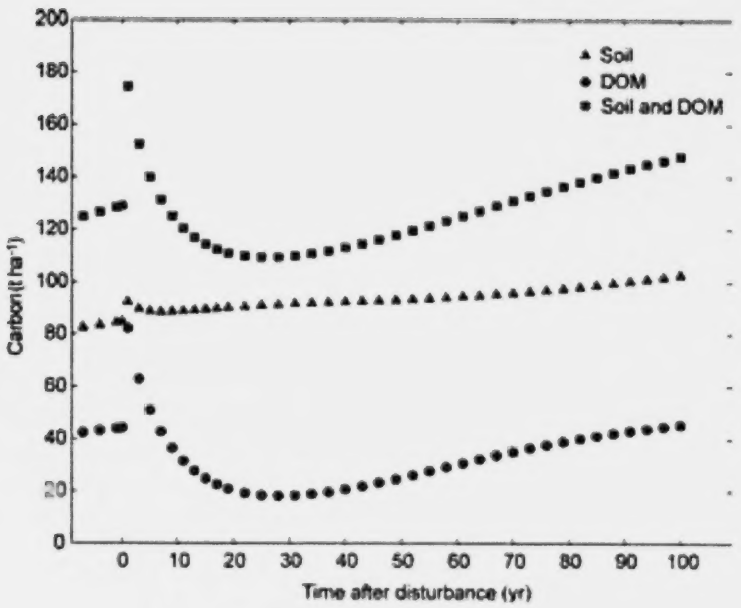

Figure 1.9.
Generalized changes in soil and DOM C pools in Plonski Site Class 2 black spruce stands following clearcutting (time 0).

additions to the soil, temperature-dependent decomposition rates, and releases from disturbances such as fire. The accumulation of DOM following a disturbance is based on assumptions about litter transfer from biomass to downed woody debris and standing dead trees following fire or insect infestation, direct losses of biomass through combustion during a fire, and biomass left on site after harvest.

We used data from Plonski Site Class 2 black spruce stands to run CBM-CFS2 and demonstrate the effects of clearcutting and forest fire on soil and DOM C. Six aboveground DOM, or litter, pools that have different decay rates (very fast: 0.5 yr^{-1}, fast: 0.14 yr^{-1}, medium: 0.037 yr^{-1}, and slow: 0.0068 yr^{-1}) are distinguished in CBM-CFS2 (Kurz and Apps 1999, Apps et al. 2000). The very fast cycling C pool consists of foliage and fine, non-woody roots shed by trees and largely (i.e., 50-95%) decomposes within 2 to 3 years. The fast soil C pool contains branches, tops, smaller sub-merchantable stems, and dead coarse woody roots that decay within 5 to 22 years. The medium soil C pool is made up of larger woody debris that decomposes within 20 to 80 years. Carbon from these three DOM pools is either released to the atmosphere (83% of total) or enters (17% of total) the slow C pool. The C in this slow pool is humified organic matter that is highly resistant to decay and decomposes in 100 to 440 years. The two remaining DOM C pools are standing dead trees and dead branch snags, which are considered to be components of the medium and fast DOM C pools, respectively.

Stand-replacing disturbance by clearcut harvesting and fire affects soil C. As shown in the modelled response of total soil C to clearcutting (Fig. 1.9), CBM-CFS2 predicts an immediate increase in soil C after harvest due to the reclassification of root systems from biomass to soil upon the death of the trees. After clearcutting, total soil C is estimated at about 90 t ha^{-1} and after 100 years this increases by about 10 t ha^{-1}. A large increase in aboveground DOM results from fast cycling litter from logging slash. In this example, the initial increase in DOM C after clearcutting is about 38 t ha^{-1}. This differs substantially from wildfire effects on DOM C pools. In the case of a stand-replacing fire, all DOM pools increase greatly, with the largest increase being in standing dead trees. In subsequent years, C is transferred to the slow cycling DOM pool as the standing dead trees fall to the forest floor.

For both types of disturbance, a decades-long recovery period of litter inputs occurs as trees and other plants are re-established. According to CBM-CFS2, the sum of the DOM and soil C pools after clearcutting decreases about 20 t ha^{-1} below pre-disturbance levels 25 years after harvest, principally due to the reduced size of the aboveground DOM (i.e., litter) pools. In the generalized case shown in Figure 1.9, DOM C recovers to pre-disturbance levels about 90 years after clearcutting.

Biomass carbon

According to CBM-CFS2, stands that regenerate on a clearcut site without planting or the benefit of advance growth follow the trends in living biomass accumulation shown in Figure 1.10 (Kurz and Apps 1999). Biomass includes the volume of wood in living tree stems, other aboveground plant stuctures (e.g., branches and foliage) and root systems. According to this model, biomass in a disturbed stand is at some point removed (i.e., trees are killed by fire or stems are harvested) and then re-accumulates through four growth phases. In the model, the natural regeneration phase (I) lasts 10 years, during which time stand biomass increases slowly and linearly from zero. Following the regeneration phase, the stand enters an immature growth phase (II), where biomass increases according to a logistic growth equation. Most of the total stand biomass accumulation occurs during this phase. Phase III is the mature growth phase, during which time stand biomass content remains relatively constant. In the overmature growth phase (IV) biomass declines gradually as stand breakup occurs. In CBM-CFS2, planting has the effect of giving an initial biomass equal to that at the end of the 10-year regeneration phase, followed by biomass accumulation at the logistic growth rate in Phase II. This therefore overestimates the amount of biomass present immediately after disturbance.

Stand volume growth curves provided by Plonski (1971) for naturally regenerated, fully stocked stands in Ontario, were used to develop biomass predictions for this report. These growth curves describe gross total (at age $\geq$ 20 years) and merchantable (at age $\geq$ 60 years) stem volumes during the immature and the start of the mature growth phases. Gross stand volumes from Plonski (1971) were converted to total stand biomass C, using a wood density value of 0.453 t m^{-3}. Merchantable stand biomass C was calculated from gross merchantable stand volumes, which are provided in 10-year intervals, using the same conversion factor. Annual gross merchantable stand C was estimated by fitting the decadal data to a sigmoidal function. This approach tended to overestimate gross merchantable volume compared to Plonski (1971) values, especially at ages younger than 50 years, where the difference peaked at 7.8 t C per hectare. Gross aboveground stand C was then converted to total above- plus belowground C (Fig. 1.11) using an expansion factor for total stand C (i.e., above- and belowground C) equaling 1.222 times gross aboveground stand C.

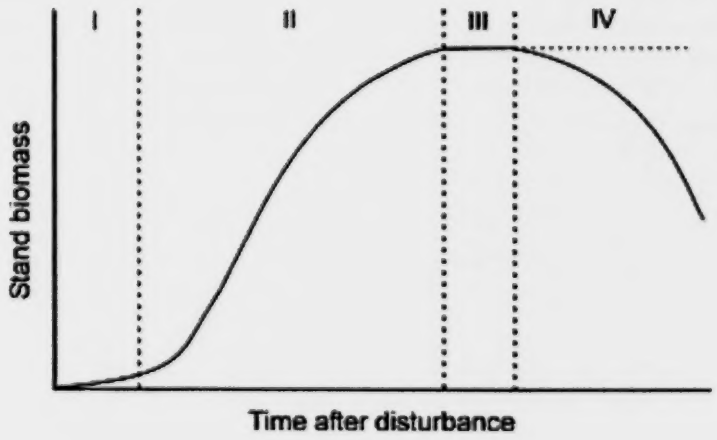

Figure 1.10.
Idealized changes in stand biomass over time since disturbance: I – regeneration phase, II – logistic growth phase, III – mature growth phase, and IV – overmature growth phase (after Kurz and Apps 1999). The dashed line in the overmature growth phase refers to stands that maintain constant biomass.

Total stand carbon

Total stand C decreases following disturbance by fire or clearcutting. This decline is faster and total stand C reaches a lower minimum value with clearcutting than with fire (Fig. 1.12). Stand C decreases by about 10 t ha^{-1} more after clearcutting than after fire disturbance. Modelled stand biomass growth is assumed to be the same for both types of disturbance. Most of the difference in total stand C between clearcutting and fire is due to loss of harvested wood from the clearcut site. In this modelled scenario, the removal of wood from the forest after harvest is considered a net C loss. Some of the C is sequestered in various types of wood products or used to provide bioenergy. By comparison, fire kills overstory trees but most of the C in the dead trees is retained on the site as a form of DOM (in this case, standing dead trees). Fire also reduces forest C by direct combustion of living biomass and dead organic matter in the litter pool. Total stand C approaches preharvest levels after about 80 years with fire and after 85 years with clearcutting. These and related values derived from Figure 1.12 are used in this report to predict total stand and biomass C at different times following disturbance.

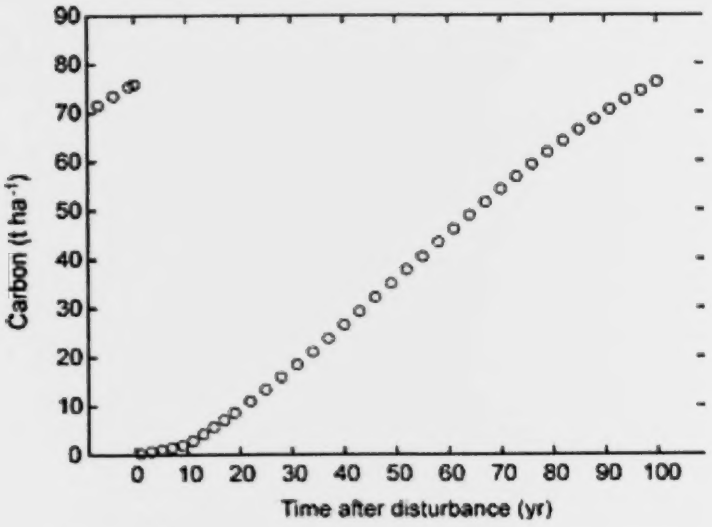

Figure 1.11.
Generalized total stand biomass C derived from Plonski's (1971) yield table for Site Class 2 black spruce.

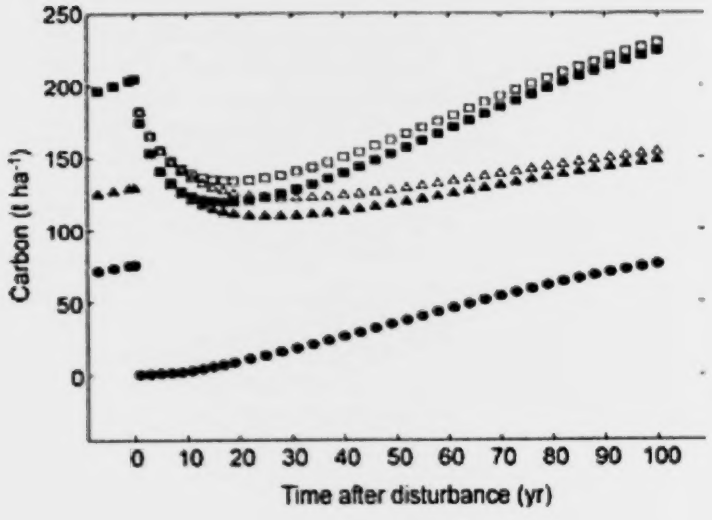

Figure 1.12.
Generalized changes in total stand (squares), soil and litter (triangles), and biomass (circles) C in black spruce stands with natural regeneration following stand-replacing disturbance by clearcutting (solid symbols) or fire (open symbols). Stand biomass for both disturbance types are equal.

2. Stand Establishment

2.1 Site Preparation

Site preparation may be defined as a mechanical, chemical, fire, or hand treatment that modifies a site to improve the establishment of a new stand of trees. Depending on the method used, site preparation enhances stand establishment by: (1) removing woody debris that could impede planting, (2) creating a seedbed where natural or direct (i.e., artificial) seeding is to be the source of the new stand, (3) reducing the growth and abundance of competing vegetation, (4) increasing soil temperature, and/or (5) alleviating soil compaction (Morris and Lowery 1988, Örlander et al. 1990, Otchere-Boateng and Herring 1990a). Because most of the site nutrient capital of northern forests resides in the forest floor, the effects of site preparation on the rates of input and decay of this layer can substantially influence short- and long-term productivity. Choice of treatments to manipulate this layer should seek to balance improved seedling establishment (i.e., short-term productivity) with conservation of organic matter to protect long-term site productivity (Prescott et al. 2000).

Site preparation techniques

Three primary types of site preparation are used in Ontario: mechanical, chemical, and prescribed burning. These techniques may be used alone or in combination, and may be applied continuously over most of a site, or only partially in strips or patches. Treatments that disturb small, isolated areas and leave most of the forest floor intact will likely influence productivity less. The area site prepared in Ontario more than doubled from about 50,000 ha in 1982 to 115,000 ha in 1990 (Armson et al. 2001). From 1998 to 2002, mechanical site preparation was most common in Ontario. It was applied to 70,000 to 80,000 ha annually, or roughly 85 to 93% of total area treated. By comparison, chemical site preparation was used on 6 to 13% of the area treated, while prescribed burning was applied to less than 1% of the area site prepared each year (CCFM 2005).

Mechanical site preparation

The initial focus of mechanical site preparation is managing logging residue, such as piling slash for burning, to facilitate subsequent treatments. Further mechanical site preparation treatments are directed at the forest floor and mineral soil surface horizons to reduce competing vegetation and create microsites tailored to the artificial and natural regeneration of selected tree species. The most common methods of mechanical site preparation are screefing (i.e., blading, scalping, root raking), trenching, inverting (i.e., mounding), and mixing, each of which may be accomplished using a variety of heavy machinery (Coates and Haeussler 1987, McMinn and Hedin 1990, Sutherland and Foreman 1995). The choice of method and equipment used is based on site characteristics and the specific objectives of site preparation relative to the crop tree species and method of regeneration selected (Coates and Haeussler 1987, Morris and Lowery 1988, McMinn and Hedin 1990, Sutherland and Foreman 1995, Ryans and Sutherland 2001).

Screefing refers to the removal or reduction of the surface organic layer and is commonly accomplished using a tractor fitted with a front-mounted blade or rake, riding freely over the ground. Alternatively, the Silva Wadell powered cone scarifier and anchor chains or tractor pads, or other machinery with rear-mounted attachments, can be used for screefing (McMinn and Hedin 1990, Sutherland and Foreman 1995). The percent disturbance to the forest floor by screefing is determined by the operator, site conditions, and machinery used and, according to Sutherland and Foreman (1995), has been observed to range from 35 to 80% (median 69%) of the area. The objective of screefing differs when applied to upland and lowland sites. On upland sites, the goal is to expose the underlying mineral soil. On lowland sites, the intent is to remove only the top layers of Sphagnum and associated moss species and slightly compact the underlying moss layers, without disturbing the mineral soil (OMNR 1997). In Ontario's Clay Belt, removal of surface organic layers and exposure of the underlying clay mineral soils impedes conifer regeneration and should be avoided (Arnup 1998).

The most intensive form of mechanical site preparation used in Ontario involves the removal of logging slash combined with a severe form of screefing that results in slash, downed wood, humus, and surface mineral soil layers being pushed into "windrows". Windrows are generally a few metres in width and height but may cover 5 to 15% of the site (Morris et al. 1983, Lundmark-Thelin and Johansson 1997). In extreme cases, the top several cm of mineral soil may be placed in windrows, equal to the displacement or removal of up to 600 t soil ha^{-1} (Glass 1976, Noble et al. 1977, Morris et al. 1983, Pye and Vitousek 1985, Tuttle et al. 1985, Ross et al. 1986, Schmidt et al. 1996). These long, linear assemblages of stumps, branches, forest floor, and soil may contain 10 times more soil by mass than woody material (Morris et al. 1993). Slash piles are generally left in place to slowly decompose or are burned (Morris and Miller 1994). Because windrowing of nutrient-rich material may degrade long-term site productivity, this method of site preparation is discouraged in Ontario (OMNR 2001d).

Trenching refers to the mechanical formation of more or less continuous furrows, or trenches, created in mineral soil, with the forest floor, slash, and low vegetation scattered to either side of the furrow (Sutherland and Foreman 1995). The most common type of trenching machinery in Ontario is the powered disc trencher pulled by a bulldozer, but drag scarifiers (e.g., anchor chains, barrels) and non-powered disc trenchers may also be used (Ryans and Sutherland 2001). The depth of disturbance is regulated by the amount of weight placed on the trenching device and is set according to the thickness of the surface to be treated. Sutherland and Foreman (1995) surveyed a number of sites in northwestern Ontario and observed that in 24 sites prepared by trenching alone, the range of disturbance to the forest floor was 32 to 80% (median 59 to 62%). Where screefing and trenching were combined, the range in forest floor disturbance was 13 to 92% (median 55%).

"Inverting" or "mounding" techniques flip a block or strip of the forest floor and soil upside down, forming raised planting beds in patches or rows distributed over a cutover area (Sutton 1993, Sutherland and Foreman 1995). This method is selectively applied to improve drainage and increase temperature on sites with cool, wet mineral soils (Morris and Lowery 1988, McMinn and Hedin 1990, Örlander et al. 1990, Sutton 1993, Sutherland and Foreman 1995). The shape, position, size, composition, and substrate of mounds varies considerably with numerous mechanical methods used to create them, which in turn have substantial effects on microsite environment and crop tree response (Sutton 1993). For example, the thickness of the mineral soil cap, if any, covering an inverted forest floor organic matter layer depends on the depth of the implement used. Sutherland and Foreman (1995) found that the least disturbance of the forest floor occurred with inverting compared to other, more broadcast forms of mechanical site preparation, and ranged from 26 to 70% (median 48%). The relatively moderate, partial mechanical disturbance by mounding and inverting treatments is less effective at assisting the growth of regenerating tree seedlings on better sites where competition is typically more intense.

Mixing refers to mechanical site preparation techniques with which fine debris and organic surface horizons are mixed, or mulched, with the underlying mineral soil using rotating drums, tillers, or other devices (McMinn and Hedin 1990, Sutherland and Foreman 1995). Mixing may be particularly beneficial in the control of competing vegetation that reproduces aggressively from rhizomes or suckers, by chopping these structures into pieces small enough to inhibit their sprouting (McMinn and Hedin 1990, Sutherland and Foreman 2000). Incorporating organic matter into mineral soil by mixing may also improve soil physical properties such as permeability and aeration, thereby increasing drainage and root growth, particularly when applied to sites with finer-textured soils (Morris and Lowery 1988, Grossnickle and Heikurinen 1989, Örlander et al. 1990, Sanchez et al. 2003). Mixing can also reduce the susceptibility of soil to compaction (Soane 1990). Mixing surface organic layers with mineral soil may also improve site nutrient retention relative to treatments that remove this layer, since it may slow the rate of nutrient mineralization, and reduce initial nutrient losses by leaching (Salonius 1983, Binkley 1986, Krause and Ramlal 1987, Morris and Lowery 1988).

Chemical site preparation

Site preparation using chemicals alone is a method that focuses on enhancing stand establishment primarily through controlling the growth of competing vegetation, as opposed to increasing soil temperature, alleviating compaction, etc. (Otchere-Boateng and Herring 1990b, Campbell et al. 2001). Chemical site preparation is a viable option when movement of heavy machinery on a site can increase soil compaction and erosion, especially on wet or clay soils or on sites with steep slopes (Otchere-Boateng and Herring 1990b, OMNR 1997, Campbell et al. 2001). This approach is also well-suited for relatively fertile sites where vigorous sprouting of residual competing vegetation is anticipated. Chemical site preparation also has the advantage of not disturbing the soil seed bank or creating mineral soil seedbed for invasive, prolifically seeding species, as may occur after mechanical methods. Since there is often a need to remove logging slash and expose mineral soil as well as control competing vegetation, chemical site preparation is commonly applied following mechanical site preparation (Sutherland and Foreman 1995, OMNR 1997, Campbell et al. 2001).

Prescribed burning

Similar to mechanical site preparation, fire may be used primarily to remove logging slash to facilitate further treatments, as well as to, for example, decrease wildfire hazard and improve aesthetics. (Feller 1982). Piling and burning of logging slash is currently applied to about 18,000 ha annually in Ontario (CCFM 1997). Burning of slash piles is commonly performed where full-tree logging results in accumulation of considerable amounts of slash at the roadside, where it may take 10 to 30 years to fully decay (Luke et al. 1993). Where this material cannot be practically returned to the cutover site, it is piled and burned as it has little value as wildlife habitat, represents a wildfire hazard, and typically occupies 4 to 7% of the harvested area that could otherwise be promptly regenerated (Luke et al. 1993, OMNR 2001d).

Alternatively, fire is used as a broadcast, more ecologically focused means of site preparation specifically tailored to the regeneration of fire-adapted tree species (Hawkes et al. 1990, McRae et al. 2001). Currently, prescribed burning is applied to only a few hundred hectares each year, but its increased use is strongly encouraged whenever it is safe and silviculturally appropriate, because it simulates the ecological effects of natural fire on forest ecosystems (OMNR 2001d). In this case, two key objectives of using fire for site preparation are to enhance regeneration by reducing the thickness of the organic layer to create planting sites or seedbed, and to destroy propagules (seeds, rhizomes) and stems of competing plants (McRae et al. 2001). The depth of forest floor disturbance depends on the heat of the fire, the type of organic layer present, and the environmental conditions at the time of the fire (McRae et al. 2001). Prescribed burning is currently used primarily in cutover areas in the northwestern region of Ontario to assist jack pine and black spruce regeneration after clearcutting (CCFM 2005). Understory prescribed fire, sometimes in combination with partial overstory harvesting, is also used to assist the natural regeneration of red pine and white pine ecosystems in Ontario (McRae et al. 2001). The intent of understory prescribed burning is to produce a surface fire hot enough to control competing vegetation without damaging overstory pine trees. On sites where stump or root sprouting of hardwood vegetation occurs, multiple burns may be needed to control this source of competition. Burning also can improve the seedbed for these and other fire-adapted tree species by removing the quick drying surface litter and exposing the underlying, nutrient-rich humus and mineral soil layers.

Site preparation effects on soil and dead organic matter carbon

Site preparation influences the amount of C contained in the DOM and mineral soil pools dependent on the fate of the forest floor (i.e., burned, buried, mixed), and the effects of this disturbance on factors and processes controlling DOM decay, microbial populations (i.e., size, diversity, and activity) and the soil environment (i.e., temperature, moisture content, aeration) (Marshall 2000). The largest relative effects of site preparation on soil C will occur through changes in the size, location, and environment of the DOM pool, and the distribution

and abundance of of soil fauna and microflora, which influence decomposition (Marshall 2000). Compared to mineral soil C pools, which change very slowly and are largely unaffected by site preparation unless treatments are unnecessarily severe, this pool is very responsive to disturbance (Johnson 1992a, Yanai et al. 2003). Obviously, mechanical methods that remove the forest floor organic layers and deposit them elsewhere (e.g., windrowing) will result in the largest reductions in forest floor and total soil C content (Johnson 1992a, Bock and Van Rees 2002), while spot treatments such as patch scarification would minimize these effects. However, the fate of this C after broadcast treatments ultimately depends on whether this material is burned or its decomposition is otherwise accelerated (Johnson 1992a).

Changes in forest floor C remaining on site following treatment are assumed to be relatively short-term, with a decline and recovery to pre-treatment levels during the first few decades after site preparation as litter C inputs from forest vegetation attain a balance with C lost through decomposition (Cooper 1983). However, within this period the effects of mechanical site preparation on the forest floor C can vary (Johnson 1992a, Hoover 2003), with positive, negative, and no effect being reported in northern forests, depending on site and soil characteristics, forest type, method used, and time after treatment (Johansson 1994, Keenan et al. 1994, Burgess et al. 1995, Örlander et al. 1996, Schmidt et al. 1996, Mallik and Hu 1997, Trettin et al. 1997, Bulmer et al. 1998, Duchesne and Wetzel 1999, Bock and Van Rees 2002, Frey et al. 2003a). For example, mechanical site preparation that mixes litter and logging slash with surface soil layers increases C in the mineral soil, but may also speed decomposition of this organic matter (Salonius 1983, Fox et al. 1986, Smethurst and Nambiar 1990, Mallik and Hu 1997, Prescott et al. 2000, Bock and Van Rees 2002, Sanchez et al. 2003). In forested wetlands, losses of soil C due to mechanical site preparation will likely be greatest when soil temperatures are elevated, and soil porosity and gas exchange are increased and may occur over much longer periods than for upland forests (Trettin et al. 1995). According to Trettin et al. (1995, based on Gorham 1991), soil C losses to decomposition in northern forested wetlands resulting from harvesting and site preparation could total from 10 to 50 t ha^{-1} and take from 40 to 120 years to be replaced. Unfortunately, information on long-term changes in surface organic C of northern forests in response to mechanical site preparation is extremely limited. However, it seems reasonable to assume that treatments that result in greater mixing and incorporation of forest floor C into the mineral soil will lessen any negative effect of mechanical site preparation on soil C (Smethurst and Nambiar 1990, Burgess et al. 1995, Örlander et al. 1996, Johnson and Curtis 2001, Yanai et al. 2003).

The intensity of a fire is the main determinant of the amount of soil C lost when prescribed burning is used for site preparation (Johnson 1992a, Johnson and Curtis 2001). Low-intensity prescribed burning typically results in a direct release of C from the DOM pool into the atmosphere coupled with increased C in the top 5 to 10 cm of the mineral soil within the first 10 years after the burn (Johnson 1992a, McRae et al. 2001). This redistribution of DOM C to mineral soil C is due in part to the formation and leaching of charcoal, a highly recalcitrant form of soil organic matter. The net result of prescribed burning is a small, transient loss or redistribution of C, with little change in total soil C over the longer-term (Johnson 1992a, Johnson and Curtis 2001, Knoepp et al. 2004). Relatively large changes in mineral soil C are likely only when fire intensity is high enough to change the physical and chemical properties of soil, such as for broadcast burning of large amounts of logging slash or when slash is piled and then burned (Johnson 1992a).

Chemical site preparation likely has very little comparative impact on DOM and soil C pools since it does not physically disturb these pools (Ross et al. 1986, Wood et al. 1989). However, slower regrowth of competing vegetation can result in elevated soil temperatures (Wood and von Althen 1993), which (depending on moisture content) could enhance decomposition of the forest floor litter layers temporarily (i.e., until crown closure occurs). Inhibition of revegetation by herbicides may contribute to greater leaching of nutrients released by enhanced decomposition and mineralization (Ross et al. 1986, Slay et al. 1987, Smethurst and Nambiar 1995). As well, reduced litter inputs to the forest floor could contribute to a reduction in the mass and C content of the DOM pool. Information on the response of decomposition to chemical site preparation is limited, but no

significant direct or indirect effects of operational concentrations of herbicides on leaf and forest floor litter mass loss rates have been reported (Fletcher and Freedman 1986, Malik and Vanden Born 1986, Marshall 2000).

Site preparation and stand biomass carbon

During the regeneration phase of stand development, percent crown cover is indicative of the relative rate of C accumulation into living and dead biomass. Prior to crown closure, NPP and NEP of a site are usually below their maximum potential rates. Site preparation can increase stand C sequestration if it reduces the time to crown closure of a new stand of trees. This can be achieved where site preparation treatments result in increased growth rates and/or survival (i.e., increasing seedling density) (Thomson and McMinn 1989). Site preparation favours the growth and survival of crop trees by improving the seedling environment, increasing resource availability, and suppressing the regrowth of competing vegetation (McMinn and Hedin 1990, Örlander et al. 1990, Sutherland and Foreman 1995, Grossnickle 2000, Ryans and Sutherland 2001, McKinnon et al. 2002). Where the forest floor layers are reduced or removed, increased soil warming has been consistently shown to have a strong positive effect on seedling growth and survival in northern forests. This can greatly shorten the regeneration phase of stand establishment (Grossnickle et al. 2001, McKinnon et al. 2002). Soil warming improves plantation establishment by effectively increasing the length of the growing season, improving shoot and root growth, and enhancing the uptake of water and nutrients. On a harvested black spruce-lodgepole pine-feather moss site in British Columbia, site preparation increased total height after 10 years by up to 30% and tree volumes tripled compared to the untreated control (Bedford and Sutton 2000). These positive growth responses were attributed primarily to soil warming following site preparation.

Since most nutrients taken up by plants come from decomposition of organic matter, the effects of site preparation on the amount, location, and decomposition of organic matter will affect nutrient availability and forest biomass accumulation (Marshall 2000). Site preparation may inadvertently reduce C sequestration if the nutrient-bearing fine woody debris, forest floor, and mineral soil surface horizons are removed, and long-term site productivity is decreased (Binkley 1986, Henderson 1995, Fox 2000). Windrowing followed by screefing is an extreme but not uncommon form of mechanical site preparation that removes the forest floor and surface mineral soil layers. This method often improves plantation establishment, but may also increase soil erosion, reduce the plantable area by up to 15%, and significantly reduce tree growth (Glass 1976, Morris et al. 1983, Pye and Vitousek 1985, Tuttle et al. 1985, Morris and Miller 1994). Decreased growth potential is associated with nutrient deficiencies that result from the displacement of nutrient-rich, organic and surface mineral soil horizons into windrows (Noble et al. 1977, Morris et al. 1983, Tuttle et al. 1985, Sutton and Weldon 1995), which may contain 10 to 20% of the site nitrogen reserves and 15 to 40% of the site phosphorous, potassium, calcium, and magnesium (Morris et al. 1983, Pye and Vitousek 1985, Fox et al. 1986). These nutrient removals are generally larger than those associated with harvesting and may not be replenished during the course of a rotation (Neary et al. 1984, Vitousek and Mattson 1985). Sites with inherently low fertility are most sensitive to these treatments (Ballard 1978, Burgess et al. 1995). Volume yield reductions of 10 to 25% in pine plantations in the southeastern U.S. were predicted to result from windrowing (Neary et al. 1984). In New Zealand, a net decline in harvestable volume of 28% was reported for radiata pine (*Pinus radiata* D. Don) due to windrowing (Dyck et al. 1989). Mechanical site preparation may also reduce seedling growth and stand establishment if the machinery used for screefing or trenching compacts the soil (Tuttle et al. 1985, Ross et al. 1986, Corns 1988, Marshall 2000), as discussed in Section 5.1.

Prescribed fire can also influence regeneration through short-term effects on competing vegetation, the seedling microenvironment, and soil biota involved in decomposition and nutrient mineralization. As for mechanical site preparation, removal or reduction in the thickness of the forest floor by fire, as well as blackening of the forest floor surface, increases soil temperature, which may improve seedling growth (Wells

et al. 1979, Weber et al. 1995, Marshall 2000, McRae et al. 2001, Frey et al. 2003a). The temporary pulse in nutrient availability released from organic matter following fire may also increase seedling growth and long-term site productivity (Wells et al. 1979, Weber et al. 1995, McRae et al. 2001, Frey et al. 2003a). However, nutrient losses to volatilization, fly-ash, and leaching associated with high-intensity fire may be substantial when much of the organic layer is consumed and revegetation is comparatively slow (Ballard 2000, Prescott et al. 2000). Although temporary reduction in competing vegetation through burning results in a transient improvement in site moisture availability, moisture levels may be reduced by relatively hot fires due to the formation of a charred, water-repellent surface that promotes runoff and impedes infiltration (Wells et al. 1979, Ballard 2000). Many beneficial soil organisms (e.g., mycorrhizal fungi) often show short-term reductions in numbers and activity after fire, particularly where burning is intense and causes large reductions in organic matter (Perry et al. 1987, Marshall 2000).

Chemical site preparation can significantly improve stand establishment and forest C storage without disrupting the nutrient capital in the forest floor organic layers, or causing any associated adverse effects on long-term growth potential (Dimock et al. 1983, Ross et al. 1986, Wood et al. 1989, Stein 1995). As well, this method of site preparation may have the least effect on the formation of mycorrhizae and other soil organisms beneficial to seedling establishment, which are sometimes reduced by slash burning and mechanical treatments that disrupt the surface organic layers (Perry et al. 1987, Marshall 2000). In a boreal clearcut, planted white and black spruce seedling survival was increased by more than 25%, height increment by more than 50%, and basal diameter by as much as 100% in the fifth year after chemical site preparation, as compared with a screefing only treatment (Wood and von Athen 1993). The effects of a single chemical site preparation treatment on crop tree performance may also be quite persistent, as demonstrated for conifer plantations in the southeastern U.S. For example, chemical site preparation that effectively controlled the early abundance of hardwood competition increased survival, total height, diameter at 1.3 m, and volume of loblolly pine (Pinus taeda L.) 27 years after treatment (Glover and Zutter 1993). Chemical site preparation also avoids growth reduction of planted seedling associated with disruption, redistribution, and/or removal of the forest floor, which is the preferred rooting horizon of certain species such as white spruce (Balisky et al. 1995, Grossnickle 2000). Wider use of chemical site preparation in Ontario to increase forest C storage should be more fully explored.

2.2 Planting Versus Natural Regeneration

Disturbed forests can be regenerated by letting them regenerate naturally, or through planting or artificial seeding. In Ontario, approximately 240,000 ha yr^{-1} have required regeneration since 1990, of which about 200,000 ha yr^{-1} have been harvested and about 40,000 ha have burned. During this time, an average of 73,000 ha yr^{-1} have been regenerated by tree planting. Of the remaining areas, about 47,000 ha yr^{-1} have received some form of treatment to encourage regeneration from seed (20,000 ha yr^{-1} by artificial seeding, 15,000 ha by seed tree, shelterwood, or strip cutting, and about 12,000 ha by selection harvesting). Approximately 120,000 ha yr^{-1} were left to regenerate naturally, i.e., they received *extensive* forest management (OMNR 1998, 2001a,b).

Natural regeneration

Natural regeneration of forest stands following disturbance arises from three primary sources: new seedlings formed by post-disturbance germination and establishment, vegetative reproduction (i.e., the "bud bank"), and advance growth (i.e., the "seedling bank") from species present on site before and surviving the disturbance (Greene et al. 1999, Groot et al. 2001). Depending on the type and severity of disturbance and the pre-disturbance stand structure and composition, one or all of these sources have a role in stand regeneration.

Natural regeneration from seeds dispersed after disturbance can be secured using several methods. Silvicultural techniques that augment either the occurrence or success of seed-based methods are known as "assisted seeding" or "planned natural regeneration". Where clearcutting or stand-replacing wildfire has occurred, natural seeding from adjacent areas of undisturbed forest can provide only limited regeneration because seed dispersal distance by wind is less than 50 m for most pines and about 80 m for black spruce (Fleming et al. 2001). The observation of seedling recruitment only near the borders of harvested or burned area and undisturbed forests undoubtedly led to the development of strip and block clearcutting, also termed modified harvesting (Jeglum and Kennington 1993). In this approach, comparatively small strips and blocks, or patches, are clearcut to retain an adjacent natural seed source and provide a microenvironment conducive to seed germination and seedling establishment. Once an acceptable level of regeneration is achieved, the uncut blocks or strips are harvested. The cut blocks or strips in these modified harvesting systems are sometimes mechanically site prepared to provide more receptive seedbeds for germination. About 1,000 to 2,000 ha yr^{-1} of black spruce and jack pine stands are regenerated in Ontario by modified harvesting (CCFM 1997). This approach has also been used successfully to promote the natural regeneration of other boreal species, such as white birch and eastern white cedar (Perala and Alm 1990, Schaeffer 1994).

The aerial seed bank and serotinous cones of jack pine and black spruce are reproductive adaptations that enable these species to regenerate naturally after disturbances (Greene et al. 1999, de Groot et al. 2004). Natural regeneration from seed released from the several cohorts of cones retained in the tree crowns will follow fire in jack pine and black spruce, provided the cones are not consumed by the fire (Greene et al. 1999, Groot et al. 2001). An average of 1.3 million viable seed ha^{-1} was released from the aerial seed bank of a 65-year-old jack pine stand after fire (de Groot et al. 2004). Jack pine and black spruce can also be regenerated by scattering the tops of harvested trees after clearcutting, with the high surface temperatures that develop on boreal clearcuts causing cones to open and release the seeds (Groot et al. 2001). Fire or scarification can create a suitable seedbed by reducing the thickness of the duff layer as well as stimulating the release of seeds from the cones of jack pine and black spruce on the forest floor (Groot et al. 2001). Seed of most other conifer species remains viable for less than 16 months and does not form a significant soil seed bank, while several hardwood tree species (e.g., cherry, basswood) may form a persistent soil seed bank that germinates following disturbance (OMNR 2000b).

Even-aged management using partial cutting systems (i.e., seed tree, shelterwood) retain a portion of the original overstory trees as a source of seed and to provide moderate shade to create a suitable microenvironment for the establishment and growth of natural regeneration. These methods are operationally used for species such as white pine, red pine, and mid-tolerant hardwood species, and occasionally white spruce or black spruce (about 12,000 ha per year, 70% of which is under shelterwood management) (CCFM 1997). For white and red pine, successful natural regeneration is dependent on partial harvesting and site preparation to create seedbed timed to coincide with years of above-average cone crops that occur comparatively more infrequently than for spruce (Horton and Bedell 1960, OMNR 1997).

Artificial, or direct, seeding is considered a form of assisted natural regeneration. Where the natural seed supply is inadequate, seeds can be distributed from aircraft, ground vehicles, or manually (Fleming et al. 2001). Direct seeding is commonly used to regenerate jack pine, but only occasionally for black spruce. Since hundreds of thousands of seed per hectare are needed to obtain adequate stocking (Cayford 1974, Fleming et al. 2001, Groot et al. 2001), this method is suitable only when the seed supply is inexpensive and plentiful.

Natural regeneration also occurs via vegetative reproduction from the bud bank, through sprouting from root systems, seedling root collar, cut stumps, or lower branches. Prolific formation of new stems through root suckering is the primary method of vegetative reproduction of poplar species (Peterson and Peterson 1996, Frey et al. 2003b). When the stems of aspen are damaged by fire or harvesting, dormant buds on roots are released from apical dominance and produce large numbers of sprouts or "root suckers" (Greene et al. 1999, Frey et al. 2003b). Because suckers often are produced in high densities and grow rapidly, they outcompete most other tree species regeneration. Aspen suckers may reach heights of 1 m in the first year and 3 to 5 m within five years (Anderson et al. 2001, Frey et al. 2003b), by far the fastest early growth rate of any boreal tree species in Ontario. The ability of hardwood tree species to initiate new stems from buds located on the basal stem or seedling root collar also provides a valuable source of natural regeneration for red oak, basswood, and other species (Dey and Parker 1996, Anderson et al. 2001). As well, sprouting from cut stumps after harvesting contributes a valuable source of natural regeneration of these species (OMNR 2000b, Anderson et al. 2001). Layering, or the vegetative reproduction from buds on the main stem and lower branches that become buried in forest floor material, provides a source of regeneration for black spruce and eastern white cedar, and infrequently for larch, balsam fir, and white spruce (Archibald and Arnup 1993, Greene et al. 1999).

Advance growth is often used to regenerate boreal conifer and mixedwood stands when sufficient numbers of the desired species are present. Young seedlings and saplings in the understory are protected during clearcut harvesting so that they can form a large part of the developing stand, greatly reducing rotation length. This approach to natural regeneration is most common in the northern boreal forest where the fairly open forest canopy permits the recruitment of a younger cohort of trees. The *careful logging with protection of advance growth* (CLAAG) harvesting method is commonly practiced in lowland black spruce forests of northeastern Ontario to take advantage of the advance growth formed by layering on these sites (Archibald and Arnup 1993, Groot et al. 2001). The establishment and cultivation of advance growth beneath a partial overstory canopy or within canopy gaps is a fundamental component of the shelterwood and selection silvicultural systems, respectively, which are used to manage mid-tolerant and tolerant hardwood species in Ontario (OMNR 2000b, Anderson et al. 2001).

Planting

Artificial regeneration by planting is more costly than natural regeneration because it requires the production and operational planting of tree seedlings as well as site preparation, vegetation management, and tending. In addition, more extensive site preparation is often required when artificial regeneration is used both to make sites accessible for tree planters and to produce suitable planting spots. Container stock now accounts for almost all trees planted in Ontario but, as recently as 1990, bareroot seedlings accounted for about half of all planting in

the province (Armson et al. 2001, Mohammed et al. 2001, Odlum et al. 2001). In general, bareroot seedlings are larger than container stock and are grown outdoors in nursery beds on a 2- to 3-year rotation. Container seedlings used in Ontario are grown in greenhouses in peat moss or peat moss mixed with vermiculite in less than 3 months.

The survival and growth of nursery-grown stock after outplanting is affected by seedling type (bareroot versus container), size, and physiological condition (Menes et al. 1996, Colombo et al. 2001, Mohammed et al. 2001). Larger seedlings generally survive and grow better than smaller ones, regardless of stock type, particularly on sites with heavy herbaceous competition (Sutherland and Day 1998, Mohammed et al. 2001). Paterson (1997) found that larger black spruce container seedlings had better stem growth than smaller container stock and McMinn (1982) observed better survival and faster growth by larger white spruce seedlings when planted on sites with intense competition. The effects of plant size and physiology on the growth of plantations can be expressed in both the short- or long-term. Arnott and Burdett (1988) reported that initial size advantages of bareroot stock may be overcome by the faster growth of container stock within a couple of years after planting. Scarratt and Wood (1988) found no significant differences in the size of container and bareroot seedlings in 9-year-old black spruce plantations. In another instance, differences in seedling size in older plantations were correlated to the initial size of the planting stock (Scarratt and Wood 1988). In general, larger seedlings will exhibit better plantation performance than smaller seedlings, other factors being equal. In some cases, the use of larger seedlings can be an alternative to the combination of small stock plus relatively intensive site preparation, thus reducing the potential adverse effects of site preparation (see Section 2.1). The competitive fitness of planting stock, and plantation establishment, can be improved by cultural means other than increasing seedling size, (e.g., nutrient loading, mycorrhizal inoculation, increased root growth potential) (Perry et al. 1987, Colombo and Noland 1997).

Regardless of stock type or seedling size, it is clear that low vigour seedlings do not perform adequately after planting. The identification and removal of damaged or low vigour nursery stock improves establishment success rates and avoids costly plantation failures (Colombo et al. 2001). Where it is necessary not only to replant but also to prepare the site a second time, the cost of obtaining successful artificial regeneration can more than double. Plantation failure also results in additional C emissions (in fossil fuels) associated with producing additional nursery stock, repreparing the site, and so on. As well, the establishment, development, and thus C sequestration of the new stand is delayed. Since 1992, a provincial Stock Quality Assessment program to test the viability and rate the relative vigour of forest tree seedlings prior to planting has significantly increased the success rate for forest plantation establishment in Ontario (Colombo et al. 2001).

Choice of regeneration method and carbon sequestration

Regeneration method influences stand C sequestration through its effect on the rate of stand establishment. More rapid completion of the regeneration phase after a disturbance increases the rate of forest C sequestration. This occurs both by reducing soil and root respiration losses and earlier capture of C in tree biomass. All other factors being equal, reliance on advance growth has the shortest regeneration phase followed by planting and seeding. This assumes that advance growth is of a size, abundance, and quality worth protecting during harvest.

In conifer forests, advance growth usually allows more rapid stand establishment and shorter technical rotations than planting because it has well-established root systems and is usually larger than planted seedlings (Hellum 1978, 1979, Korpela et al. 1992, Morin and Gagnon 1992, Archibald and Arnup 1993). This source of regeneration typically responds positively to overstory release, depending for example on age and crown size, and has the advantage of being taller than competing vegetation in the emerging understory canopy (Ruel et al. 2000). In mature black spruce stands on wet, nutrient-poor sites, 80% of the advance growth is formed by layering. If adequately protected during harvest, released layer-origin black spruce grows

as well or better than planted stock or seed-origin natural regeneration (Vincent 1964, Morin and Gagnon 1992, Pothier et al. 1995). Although smaller stems have greater relative height growth rates after release, the larger stems retain their dominance and have greater relative volume increment over a rotation (Morin and Gagnon 1992, Archibald and Arnup 1993, Pothier et al. 1995). When abundant, vigorous black spruce advance growth is present on lowland sites in northeastern Ontario, this regeneration can provide the bulk of the future stand and rotations can be reduced by as much as 10 to 30 years compared to planting (Archibald and Arnup 1993, Groot 2001).

Natural regeneration from seed is the slowest form of forest renewal. It requires the parallel occurrence of abundant viable seeds, sufficient suitable seedbeds for germination, and environmental conditions favourable for seed germination and the survival and growth of the seedlings. These conditions are unlikely to occur in managed stands without careful planning and significant silvicultural intervention (Greene et al. 1999, Groot et al. 2001). The small size of new germinants makes them especially susceptible to damage from drought, herbivory, mechanical damage by leaf litter, and the extreme temperatures that occur near the soil surface, with first-year mortality being relatively high (Arnott 1974, Fenner 1987, Greene et al. 1999). Despite dispersal of a few million viable seeds per hectare by conifer species during years of bumper crops, < 10% of these seeds give rise to successfully established seedlings the following growing season (Greene et al. 1999). The success of seed-based regeneration depends on the production of overwhelming numbers of germinants, only a small percentage of which survive as individuals in a new stand (Fig. 2.1). Ironically, where seed is plentiful and conditions for seed germination and germinant survival are favourable, stocking in the new stand can be extremely dense, resulting in a period of self-thinning and growth stagnation. This is most common in naturally regenerating jack pine, which requires precommercial thinning to avoid a prolonged period of intense intra-specific competition, stand growth reduction, and self-thinning (see Section 3.2).

Planted seedlings grow more rapidly than those regenerating at the same time from germinating seeds, especially in conifers, due primarily to the large size of nursery stock at the time of planting (Fleming et al. 2001). Immediately after germination, a conifer seedling consists of a short stem, usually not more than 2 to 3 cm in length, and fewer than 7 cotyledons. Much early growth is directed to establishing a root system. At the end of the first year, naturally regenerated conifer seedling shoots mostly consist of 10 to 20 needles on a stem that may be only 3 to 4 cm tall. Height growth of not more than 1 to 3 cm yr^{-1} is common in naturally regenerated conifers in northern forests for the first 3 to 5 years after germination (Wood and Jeglum 1984, Fleming et al. 2001). Conifer nursery stock is several times larger than naturally regenerated trees at the time of planting. The near ideal growing conditions in greenhouses result in container seedlings that are often

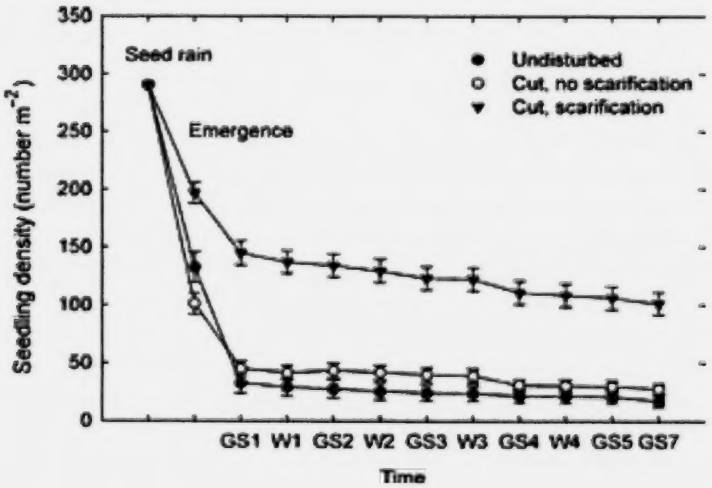

Figure 2.1.
Trends in mean seedling density of naturally regenerated white pine seedlings over time in undisturbed (solid circle) and shelterwood stands that received no scarification (open circle) or scarification (solid triangle). About 290 viable seed m^{-2} were hand sown in 9 to 12 plots per treatment. Seedling emergence and mortality were assessed after each growing season (GS) and overwinter (W) period for seven years after sowing (Parker, unpubl. data).

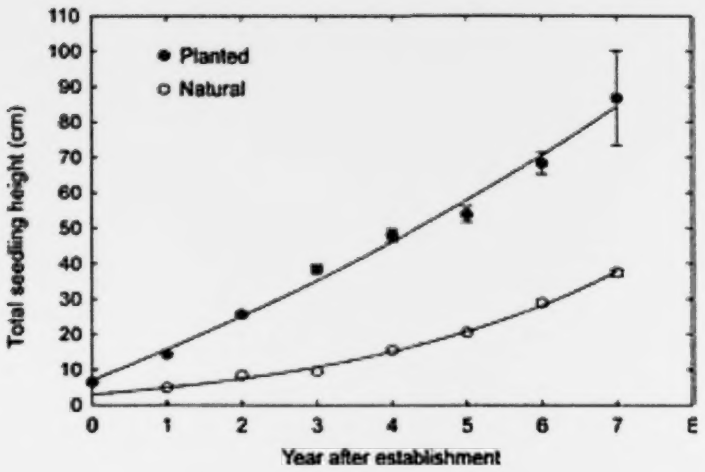

Figure 2.2.
Mean (± 1 standard error) total height for naturally regenerated and container-grown white pine seedlings seven growing seasons after establishment in a central Ontario shelterwood. Height at year 0 represents the height of container stock at the time of spring planting (Parker and Morneault, unpubl. data).

as much as 30 cm tall when planted (although 15 to 20 cm is more common) with hundreds of needles and several lateral shoots. First-year height growth of planted container and bareroot seedlings is expected to be 10 cm or more, with the rate of height growth accelerating annually. A plantation established with vigorous nursery stock is expected to consist of trees taller than the competing vegetation within 5 years for jack pine and 7 years for black spruce. Differences in early growth rates likely translate to growth advantages of 5 to 10 years for planted trees over trees established at the same time from seed (Groot 2001) (Fig. 2.2). Planting stock may also have other growth advantages over natural regeneration if genetically improved seed is used or if nursery culture (e.g., mycorrhizal inoculation, nutrient loading) is used to produce faster growing, more competitive seedlings (Colombo and Noland 1997).

Comparative carbon storage in planted and naturally regenerated stands

In CBM-CFS2 (Kurz and Apps 1999), planting results in a 10-year gain in biomass accumulation compared to natural regeneration from seed (Fig. 2.3). In our simulation of planting versus natural regeneration using CBM-CFS2, biomass accumulation is the same regardless of disturbance type. Differences occur in sizes of DOM and soil C pools with time between harvested and burned stands. Immediately after a stand-replacing fire, DOM C increases, since the standing dead trees and their roots systems are reclassified from biomass to DOM. DOM shows a similar but slightly smaller increase after clearcutting compared to fire because clearcutting removes merchantable stems for processing into wood products, reducing the amount of C in the forest. Most of the large pulse of DOM generated at the time of disturbance decays within about 20 years. About 30 years following disturbance, non-biomass C inputs to the soil and DOM pools increases annually (Fig. 2.4).

Planting results in a significant increase in total stand C compared with that obtained using natural regeneration (Fig. 2.5) as a result of the higher accumulation rates for both biomass and DOM C in plantations. The net gain in total stand C between regeneration methods differs only slightly by disturbance type. The maximum difference in total stand C between planted and naturally regenerated stands after disturbance by fire is almost 21 t ha^{-1} (Fig. 2.6). Following clearcutting, plantations sequester as much as about 20 t more C ha^{-1} than naturally regenerated stands (Fig. 2.7). The net gain in total stand C obtained by planting as opposed to natural regeneration is greater in clearcuts than in burns during the first 10 years after disturbance (Fig. 2.8). By year 11, however, burned and planted stands have the greatest net gains total stand C (Fig. 2.8).

Estimated gains in C based on tree planting between 1990 and 2012 are shown in Table 2.1. Gains in stand C are simulated based on the assumption that 90% of the stands planted in each year are regenerated successfully (with 10% of the planted sites reverting to natural regeneration). Furthermore, 50% of the area

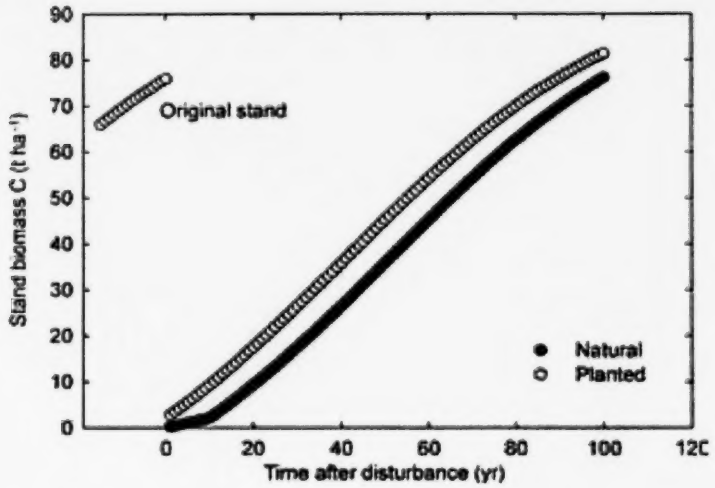

Figure 2.3.
Comparison of simulated total above- and belowground biomass C in Site Class 2 (Plonski 1971) black spruce stands regenerated by planting or natural regeneration after clear-cutting.

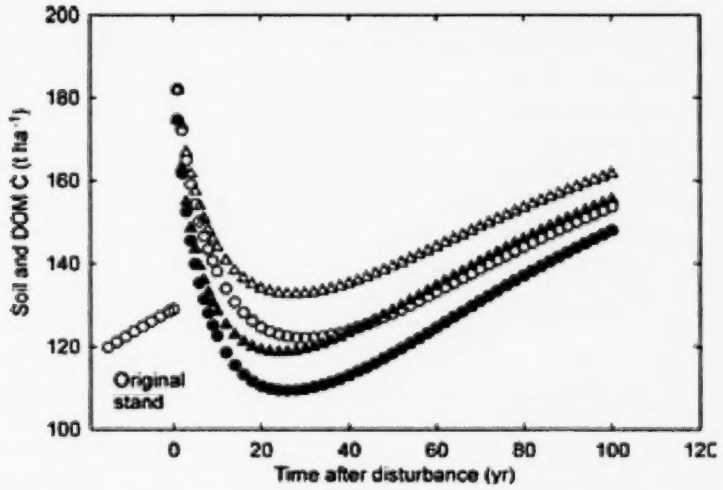

Figure 2.4.
Comparison of simulated C in soil and DOM pools (soil, litter and standing dead trees) in planted (Δ) and naturally regenerated (o) Site Class 2 (Plonski 1971) black spruce stands following fire (solid symbols) and clearcutting (open symbols).

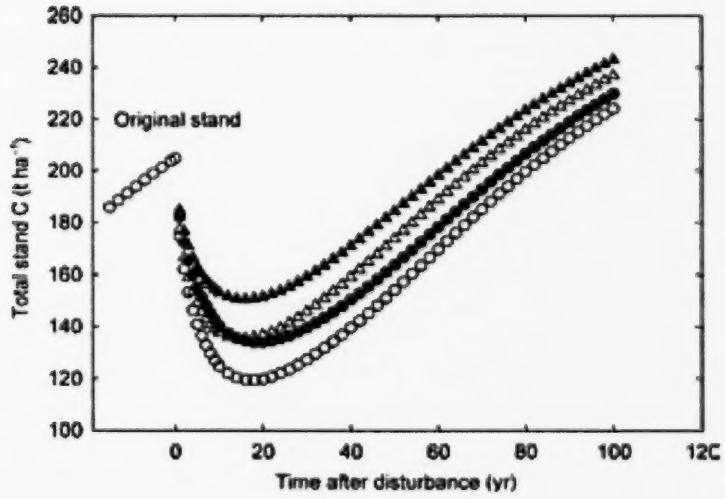

Figure 2.5.
Comparison of simulated total stand C content in planted (Δ) and naturally regenerated (o) Site Class 2 (Plonski 1971) black spruce stands following fire (solid symbols) and clearcutting (open symbols).

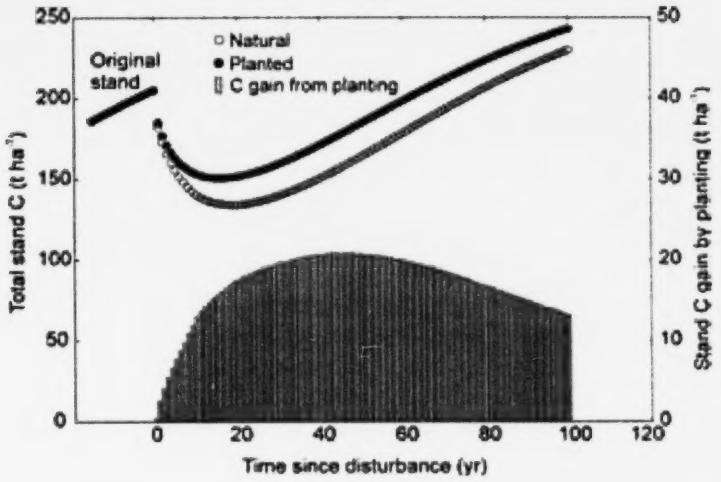

Figure 2.6.
Comparison of simulated total stand C after stand-replacing fire followed by natural regeneration or planting of a Site Class 2 (Plonski 1971) black spruce stand, and the difference between naturally regenerated and planted stands (bars).

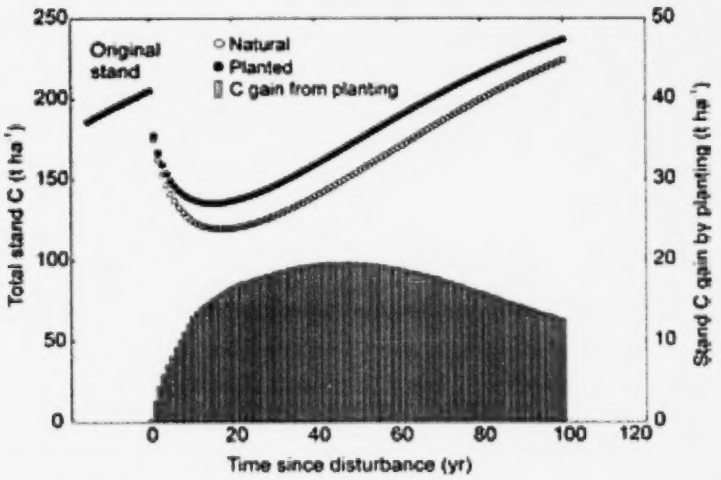

Figure 2.7.
Comparison of simulated total stand C after clearcutting followed by natural regeneration or planting of a Site Class 2 (Plonski 1971) black spruce stand, and the difference between naturally regenerated and planted stands (bars).

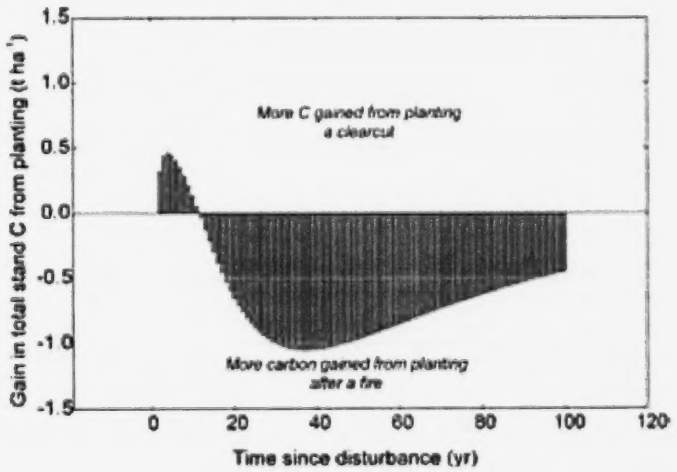

Figure 2.8.
Simulated difference in net gain in total stand C obtained by planting Site Class 2 (Plonski 1971) black spruce stand after clearcutting or a stand-replacing fire.

planted each year is assumed to be clearcut and 50% burned. The total gain in C in Ontario in 2012 and 2070 from planting compared with natural regeneration is estimated based on these assumptions and the stand C profiles shown in Figures 2.6 and 2.7 (Table 2.1).

Stands planted in Ontario in 1990 are predicted to increase C storage by 1.135 Mt (1.135 million t) by year 2012 compared to natural regeneration (Table 2.1). In 2012, the C gain from all planting since (including) 1990 is estimated to be 16.07 Mt. By 2070, the gain in C from planting between 1990 and 2012 would be 27.58 Mt (Table 2.1). For trees planted in 1990 to 1992, the C gain from planting is greater in 2012 than it is projected to be in 2070 because planted stands reach maturity more quickly than naturally regenerated stands. With maturity comes reduced growth and lower rates of C sequestration. In comparison, the more slowly maturing naturally regenerated stands will, for a period of time, have greater growth rates than planted stands. This decreases the difference in C storage between natural and planted stands.

Table 2.1. Estimated gains in C from tree planting in Ontario between 1990 and 2012[1], projected to 2070.

Year planted	Area planted (ha)	C gain from planting (t)	
		Gain in 2012	Gain in 2070
1990	71,189	1,135,322	1,053,985
1991	74,849	1,178,737	1,119,996
1992	66,693	1,034,929	1,008,428
1993	62,801	959,414	959,442
1994	56,182	838,348	872,175
1995	64,283	941,614	1,008,002
1996	72,566	1,034,991	1,142,653
1997	73,907	1,030,471	1,181,895
1998	84,704	1,151,356	1,360,084
1999	92,956	1,226,545	1,506,849
2000	81,716	1,042,565	1,344,502
2001	72,895	893,023	1,203,862
2002	72,895	854,052	1,214,850
2003	72,895	787,791	1,225,740
2004	72,895	721,956	1,229,939
2005	72,895	655,104	1,240,567
2006	72,895	587,530	1,251,130
2007	72,895	Negligible	1,255,000
2008	72,895	Negligible	1,265,235
2009	72,895	Negligible	1,275,338
2010	72,895	Negligible	1,278,750
2011	72,895	Negligible	1,288,492
2012	72,895	Negligible	1,291,543
Total	**1,206,562**	**16,073,748**	**27,578,458**

[1]Assumptions are: (1) Fifty percent of the area planted in each year was clearcut and 50% was disturbed by fire; (2) The per hectare gain in C from planting versus natural regeneration is based on the stand C values shown in Figures 2.6 and 2.7; (3) Plantation success rate was 90%, the other 10% of the area planted reverted to natural regeneration; (4) The area planted was estimated from numbers of seedlings planted each year, assuming 2,000 trees planted per hectare; (5) Areas planted for 2001 to 2009 are predicted from the average area planted from 1990 to 2000; (6) No stands were lost prior to 2070 due to natural disturbance or harvesting; (7) The C values are based on all areas planted with black spruce on Site Class 2 sites (Plonski 1971).

2.3 Vegetation Management

Forest disturbance initiates the establishment of plant communities, the composition and succession of which are strongly dependent on the type and intensity of disturbance. This vegetation can hinder the development of young forest plantations (Stewart et al. 1984, Walstad and Kuch 1987) by competing for light, water, and nutrients, i.e., all the resources needed to support vigorous crop tree growth (Cannell and Grace 1993, Nambiar and Sands 1993, Wagner et al. 2001). Competing plants can also influence the growth of regeneration positively or negatively by modifying the physical environment (e.g., air and soil temperature, humidity) of the crop tree (Jobidon et al. 1998, Cater and Chapin 2000) and by facilitating or discouraging herbivory or damage by insects and diseases (Lauer et al. 1993, Lautenschlager 1993, Périé and Munson 2000). Certain competing plant species produce growth-inhibiting chemical compounds that can adversely affect crop tree performance (Fisher 1980, Mallik 1987, Dolling 1996). Crushing or mechanical damage of conifer seedlings by neighbouring woody and herbaceous vegetation also occurs (Dolling 1996, Wagner et al. 2001). At the same time, herbaceous and woody vegetation are critical components of forest ecosystems. As such, vegetation management efforts should balance plantation establishment and growth objectives with minimizing any associated negative effects on ecosystem composition, structure, and function.

In this report, vegetation management refers to activities directed at removal of interspecific competition to favour the growth of crop trees after these individuals have become established naturally or artificially. The benefits of site preparation as a competition control measure applied prior to planting or seeding (natural or assisted) are well documented (see Section 2.1). As well, we focus our discussion on conifer species as crop trees, given their ecological and economic importance to Ontario.

Methods of vegetation management

In Canada, the primary competing species in recently disturbed areas are shade intolerant perennials, well adapted for rapid colonization, resource extraction, and short-term dominance (5-20 years) (Wagner et al. 2001, Thompson and Pitt 2003). Vegetation management is generally directed towards reducing the abundance of these species, but tailored to the specific mixture of species present by using (1) herbicides, (2) motor-manual methods (e.g., chain saws, brush saws, girdling tools), (3) mechanical treatments, or (4) biological controls (e.g., livestock, pathogens). Broadcast application of herbicides by air and ground-based machines are the most common methods presently used in Ontario, although approaches vary regionally and with forest type (Table 2.2). Although five herbicides are registered for use in forests in Canada, glyphosate accounts for more than 90% of total herbicide used over the past decade. Biological control agents such as *Chondrostereum purpureum*, a fungal pathogen used to control resprouting of broadleaf woody vegetation, are not widely used or are in the research and development stage (Thompson and Pitt 2003).

In Ontario, and Canada in general, most vegetation management efforts (>75%) are applied a few years after plantation establishment, despite the substantial benefits associated with immediate, proactive treatment (Wood and von Althen 1993, Wagner 2000, Sutherland and Foreman 2000, Campbell et al. 2001, Thompson and Pitt 2003). The lack of early vegetation management (other than that achieved by site preparation) on most harvested areas is evidenced by the annual average area tended, which approximates 37% of the average area harvested each year (Table 2.2). Vegetation management is currently evolving towards a more integrated approach that seeks to use ecologically sound, economical, and socially acceptable treatments to achieve some preferred balance of crop and weed species (Thompson and Pitt 2003). More effective competition control and larger growth responses can be obtained when vegetation management strategies are coupled with preventative approaches such as alternative harvesting systems (Lieffers and Stadt 1994, Dey and MacDonald 2001), nursery culture to produce larger, vigorous planting stock (Timmer 1997, Sutherland and Day 1998, Jobidon et al. 2003), earlier planting (Wood 1990, Pitt et al. 1999), and site preparation (Wood and von Althen 1993, Sutherland and Foreman 2000). These integrated approaches can also reduce herbicide use, a contentious issue discussed by Wagner et al. (1998, 2001)

Vegetation management and plantation productivity

Conifer seedlings consistently exhibit increased wood volume, growth, and survival following treatments to control herbaceous and woody forest vegetation (Stewart et al. 1984, Walstad and Kuch 1987, Wagner et al. 2004). Generally, early competition control focused on herbaceous vegetation, followed by control of woody vegetation maximizes conifer production (Miller et al. 1991, Wagner et al. 2001). A recent review of the longest-term (10-30 years) field studies in southeastern, northern, and Pacific northwestern forests of North America indicates that wood volume yield gains of 30 to 300% consistently result from effective vegetation management of forest plantations for a range of site conditions (Wagner et al. 2004). Major commercial tree species of northern forests exhibited wood volume gains of 50 to 450% over this period (Wagner et al. 2004).

At the stand level, vegetation management effects on crop tree growth and plantation productivity are determined in part by the differential sensitivity of stem diameter growth, height growth, and seedling survival to competition-induced resource limitations. Stem diameter is the most sensitive of the three seedling performance variables, presumably because allocation of photosynthate to cambial growth is a relatively low priority when seedlings are exposed to resource limitations due to competition (Lanner 1985). Competition reduces diameter growth proportional to total abundance of competing vegetation and is due primarily to belowground factors (Wagner 2000). By comparison, height is maintained under higher levels of competition since plant resources are preferentially devoted to attaining a competitive canopy position. Reduced height growth in response to competition does not generally occur until significant shading by overtopping woody vegetation begins (i.e., generally when the height of competition is >125% of crop tree height). The differential sensitivities of stem diameter and height growth result in a negative exponential relationship between size (e.g., stem volume, basal area) and amount of competition (e.g., stem density, leaf area, percent cover). Significant levels of seedling mortality do not occur until competition becomes relatively severe, and seedlings have exhausted their genetic potential for growth plasticity in response to an increasingly unfavourable environment. At the stand level, growth and survival reductions by competition alter the frequency distribution of regeneration, such that the proportion of larger crop trees is decreased where competition levels are higher (MacLean and Morgan 1983).

As a result of different responses of individuals to stress, and their corresponding growth and survival rates, the intensity of vegetation control can affect early stand development of plantations and the rate of C

Table 2.2. Average area (ha) treated by various competition control treatments in MNR's Northwest, Northeast, and SouthCentral Regions from 1995-2000 compared with areas harvested, site prepared, and artificially regenerated (OMNR 1998, 2000a, 2001a,b,c, 2002b)

Treatment	MNR Region			
	Northwest	Northeast	South-Central	Total
Vegetation management	27,659	47,782	2,419	78,075
Chemical – aerial	25,218	45,269	445	70,932
Chemical – ground	2,074	1,497	574	4,145
Manual	341	945	1,393	2,893
Mechanical	26	71	7	105
Harvest	82,495	107,190	20,455	210,230
Site preparation	47,697	36,230	1,807	85,778
Artificial regeneration[1]	52,450	45,695	2,533	100,678

[1]Planting and direct seeding

storage in crop trees during a rotation (Fig. 2.9) (Wagner 2000, Wagner et al. 2001). Carbon storage rates and productivity of crop trees can be maximized by vegetation management treatments that maintain woody and herbaceous competition at less than 10 to 20% cover. Less intensive treatments to prevent overtopping of crop trees by woody vegetation will sustain competitive rates of height growth, but only 20 to 30% of maximum growth potential and C storage may be achieved. Weed control efforts focused on survival require maintaining moderate densities of competition and avoiding prolonged periods of overtopping by woody species, but are associated with substantial reductions in productivity and C storage and longer rotations. With respect to the relative competitiveness of woody as compared with herbaceous vegetation, the largest reductions in stem volume growth occur when both types of competition are present, with average volume being <10% of its maximum when neither form of competition is controlled. Removal of woody vegetation can increase volume growth to 20 to 25% of maximum, while herbaceous weed control is needed to achieve the remaining 75 to 80% (Wagner et al. 2001).

Because initial differences in seedling growth can significantly influence long-term stand development, early and prolonged competition control will maximize productivity of conifer plantations (Wagner et al. 1999, Wagner 2000). Immediate, post-planting vegetation control significantly increased diameter growth of white, black, and Norway spruce (*Picea abies* L. Karst) seedlings over that exhibited when release was applied one year or later after planting (Lund-Høie 1984, Wood and von Althen 1993). The potential growth loss associated with delayed competition control increases with the length of delay, such that applying control only a few years after planting typically results in several years of future growth loss (Lund-Høie 1984, Lauer et al. 1993, Wagner et al. 1999, 2001). Moreover, later application of treatments must be more intensive and prolonged to achieve the same growth response as immediate or comparatively early treatment (Wagner et al. 1999). Obviously, prolonged competition control treatments will maximize conifer growth, although they may not be economically viable nor operationally desirable (Lauer et al. 1993, Wagner et al. 1999). For example, in Ontario five years of herbaceous vegetation control from the time of planting increased stem volume of white pine, black spruce, red pine, and jack pine seedlings by more than 600, 500, 400 and 300%, respectively, relative to trees receiving no weed control (Wagner et al. 1999).

Although more expensive and less effective in optimizing growth and wood production of conifers, even delayed release (i.e., >15 years) can significantly increase growth (Buckman and Lindgren 1962, Berry

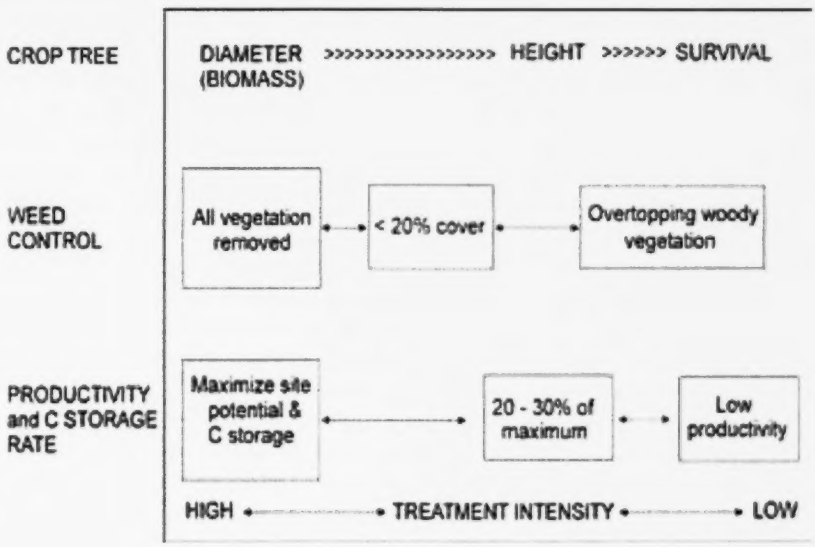

Figure 2.9. The general relationship between crop tree growth objective, level of weed control, and level of stand productivity and C storage achieved (adapted from Wagner 2000).

1982). The capacity of conifers to survive in a suppressed state and exhibit a strong growth response varies among species and with age, size, crown architecture, and other morphological features (Perala 1982, Sutton 1992, Ruel et al. 2000). For example, white pine up to 40 years old with live crown ratios greater than 30% show positive growth responses to release from overtopping vegetation (Wendel and Smith 1990). Increased growth following partial cutting and natural release has even been reported in old-growth forests of several tree species (Abrams and Orwig 1996, Latham and Tappeiner 2002, Bebber et al. 2004). For example, a more than 470% increase in average 10-year diameter growth rates was observed in 130-year-old white and black spruce after thinning (Crossley 1976).

Vegetation management and stand carbon balance

Vegetation management has little if any effect on inherent site productivity because it does not change organic matter reserves, soil nutrient pools, or significantly alter soil physical properties and normal hydrologic and microbiological function (Fox 2000, Johnson and Curtis 2001). Instead, vegetation management affects forest C balance primarily by increasing the rate of biomass accumulation and C storage by desired crop tree species. This rate is strongly dependent on the timing, intensity, and duration of competition control (Fig. 2.9). This reallocation of biomass from competing vegetation to crop trees does not influence the total stand biomass or C, but a significant reduction in the technical rotation of the crop tree species is achieved (Fleming and Wood 1996). Early, intensive vegetation management also reduces the stand initiation phase or "regeneration lag" (Kurz and Apps 1999, Lauer et al. 1993, Oliver and Larson 1996). Even when treatments are significantly delayed (e.g., >15 years), the technical rotation of crop trees can be reduced, while total living biomass C is unchanged or slightly reduced relative to undisturbed stands (Johnson 1986, Stiell et al. 1994). For example, releasing suppressed white spruce advance growth up to 75 years old from an aspen overstory can shorten the rotation by 40 to 50 years (Johnson 1986).

Vegetation management effects on soil and DOM C are short-lived and primarily occur as a result of changes in the balance between litter inputs to the forest floor and the rate at which this surface soil organic matter decomposes (Facelli and Pickett 1991, Harrington and Edwards 1999, Périé and Munson 2000). Where mechanical or manual methods are used to control woody vegetation, dead biomass is left on site as slash and becomes part of the forest floor. With chemical tending, dead aboveground biomass remains intact for some time, and gradually falls to the forest floor. Root biomass of competing species treated by either method is also added to the soil C pool. However, the reduction in competing vegetation results in a short period of lower litter inputs to the forest floor. At the same time, increased insolation of the forest floor results in elevated soil temperatures and enhanced rates of root respiration and decomposition (Polgase et al. 1992, Wood and von Althen 1993, Périé and Munson 2000, Thibodeau et al. 2000). The pulse of organic matter input to the DOM C pools after tending also provides a readily available energy source (i.e., C) and stimulates the activity of soil microorganisms involved in decomposition (Munson et al. 1993, Thibodeau et al. 2000). Enhanced decomposition and lower litter inputs slightly reduce soil C pools for a short period. However, as crown closure proceeds and the forest floor becomes increasingly sheltered from direct radiation, soil temperatures, soil microbial activity, and soil C loss from decomposition and root respiration decrease. As well, the enhanced growth of crop trees and eventual regrowth of competing vegetation result in recovery of above- and belowground litter inputs to pre-treatment levels within a few years.

Vegetation management and carbon storage in Ontario's forests

Vegetation management is an integral part of planting success on sites where competition is severe. Delayed or insufficient vegetation management treatments can significantly reduce growth in plantations and, where competition-induced mortality of crop trees is high, the area may revert to natural regeneration. In CBM-CFS2 model, planting is assumed to reduce the regeneration phase by 10 years compared to natural

regeneration (Kurz and Apps 1999). Given this assumption, planted stands receiving no competition control will exhibit reduced growth and survival and in cases of severe competition will have soil, DOM, and biomass C accumulation rates similar to naturally regenerated sites. Based on these assumptions, areas planted to black spruce in 1990 have an additional 12.6 t C ha^{-1} in biomass and 13 t C ha^{-1} in soil and DOM in 2012 when compared to naturally regenerated black spruce stands. For jack pine plantations, the total net C gain after 22 years is 23 t ha^{-1}.

Using the figures for net C gain presented above, the influence of vegetation management on forest C storage can be illustrated by the following scenario. Assume that 80,000 ha of boreal forest cutover land received a vegetation management treatment and successfully regenerated to black spruce (48,000 ha) and jack pine (32,000 ha) by planting in 1990. By 2012, the black spruce plantations store an additional 1.23 Mt C and jack pine an additional 0.74 Mt C relative to equivalent naturally regenerated stands. The total gain in C storage over the 80,000 ha is 1.97 Mt C. If in 1990, 25% less area of black spruce plantations received competition control, C storage would be further decreased by 0.31Mt C by 2012.

3. Growth-Enhancing Forest Management Practices

3.1 Genetic Improvement

Increasing the genetic potential for growth is the primary objective of tree breeding in Ontario (Joyce et al. 2001). Tree breeding is intended to provide an economic advantage to forest-dependent industries and communities by increasing the supply of wood. Genetic gains in growth potential can also increase forest and forest product C storage through reduced technical rotations, increased volume at rotation age, and/or earlier commercial thinning (McInnis and Tosh 2004). Progeny tests combined with seed orchards are the primary method of genetic improvement in Ontario (Joyce et al. 2001). Progeny tests are established using seed collected from trees in unmanaged forests that have a promising appearance ('plus' trees). Seeds from plus trees are used to produce nursery stock that is planted together to judge their relative performance. Poorer-performing half-sib families (i.e., individuals with the same seed-bearing parent but different pollen parents) are removed and the best performing populations are retained to provide seed for reforestation. About 450 ha of jack pine and black spruce seed orchards existed in Ontario in 1997 (Joyce et al. 2001). Second-generation jack pine orchards (i.e., selections of the best progeny and parents from the original progeny tests) were first established in 1998 in Ontario and will begin producing seed by about 2010 (D. Joyce, 2001, OMNR, pers. comm.).

More intense roguing to remove inferior families in progeny tests and in early generations increases the level of genetic improvement but also reduces the size of the breeding population more quickly (Namkoong 1986). This decreases genetic variability, which in turn reduces the ability of these selected populations to respond to changing ecological conditions, such as those expected to occur with climate change (Namkoong 1986). Smaller breeding populations also may reduce growth in future generations as a result of inbreeding depression (Wright 1976). While the introduction of new individuals to the breeding population would increase genetic variability and reduce inbreeding, it may jeopardize gains in growth rate achieved through prior selections. The ability to improve growth through several generations is thus constrained by the opposing goals of maintaining genetic diversity while increasing growth rates.

In Ontario, genetic improvement is realized by planting nursery stock grown from seed from first-generation seed orchards. Currently, jack pine, white pine, black spruce, and white spruce constitute most of the operational seed orchards in Ontario (Joyce et al. 2001). Height growth is improved by 3 to 8% in first-generation jack pine seedling seed orchards (Joyce et al. 2001). Some estimates suggest that gains of 10% can be achieved by increasing the intensity of roguing in seed orchards (D. Joyce, 2001, OMNR, pers. comm.). Height gains of 14% and stem volume gains of 30% at age 40 were estimated from first-generation selections of jack pine in New Brunswick's tree improvement program (McInnis and Tosh 2004). These estimates are also within the bounds of the 6 to 20% gains that have been achieved in first-generation pine breeding programs in the southeast U.S. (Pait et al. 1991). In a recent meta-analysis of growth gains accrued from forest tree improvement of four North American conifer species, gains in relative height for first generation selections of black spruce, white spruce, and jack pine at 40 to 50 years were estimated at 13%, 20%, and 28%, respectively (Newton 2003). Genetic improvement programs for other commercially important forest tree species, such as aspen, sugar maple, and white birch are non-existent or in formative stages, and will require 20 years from their inception to provide improved seed.

Genetic transformation of tree species through the insertion and controlled expression of foreign genes or modified endogenous genes is the most sophisticated method of genetic improvement available. The introduction of foreign genes has been used to increase herbicide tolerance and resistance to insects, diseases, and various forms of abiotic stress, while efforts to alter the expression of existing genes have been targeted at improving fibre quality (e.g., lower lignin content) and quantity (i.e., increased growth rate) (Mullins and Bertrand 1998, Matthews and Campbell 2000, van Frankenhuyzen and Beardmore 2004). Proponents of transgenic trees argue that use of this form of genetically improved plant material in plantation forestry can reduce harvesting

pressures on natural forests, reduce the costs associated with growth and protection of these plantations, and, in the case of improved fibre quality for example, decrease the cost and use of hazardous chemicals in the pulping process (Kaiser 2001, Mann and Plummer 2002). Critics of this technology counter that the risk of potential negative ecological effects is too large to justify the release and commercial use of this material (Kaiser 2001). At present, scientific uncertainty surrounding these adverse effects has greatly limited the field testing needed for regulatory approval and operational use of transgenic trees in forestry (Mullin and Bertrand 1998, van Frankenhuyzen and Beardmore 2004).

Genetic improvement and C storage in Ontario's forests

To calculate the increase in C storage achieved through genetic improvement, we compared stand biomass C dynamics of plantations established by natural regeneration, planting, and planting with genetically improved nursery stock. We assumed that trees derived from open-pollinated, first-generation seed orchards yield an average 5% increase in stand volume and stand biomass C up to age 80 (Fig. 3.1). After age 80, the volume and C of genetically improved stands would increase less rapidly and reach a final biomass determined by site productivity and equal to the same maximum biomass as non-improved seed sources. However, maximum biomass production would be reached sooner after planting than after natural regeneration and sooner yet in genetically improved stock.

The increased growth resulting from genetic improvement could be used to shorten rotation lengths. Alternatively, keeping rotation lengths as is would sequester additional C and provide greater timber volumes for harvest in the short term. Thus, the effects of genetic improvement on C sequestration are simulated for black spruce stands using CBM-CFS2 under three scenarios: (1) increasing C accumulation in 2012 in genetically improved stands established between 1990 and 2009 (i.e., C gain at the end of the first Kyoto commitment period), (2) using genetic improvement to increase volumes harvested and C sequestered by maintaining current rotation length (e.g., 80-year rotations), and (3) shortening the rotation of genetically improved stock to harvest wood volumes equivalent to those of naturally regenerated stands. While a 5% increase in biomass C accumulation would be accompanied by increased soil and litter C inputs, for this simulation we assume that soil and DOM C pools are unaffected. This probably only slightly underestimates the gain in total stand C due to genetic improvement, as effects on soil and DOM C pools during the first few decades after stand establishment are minor.

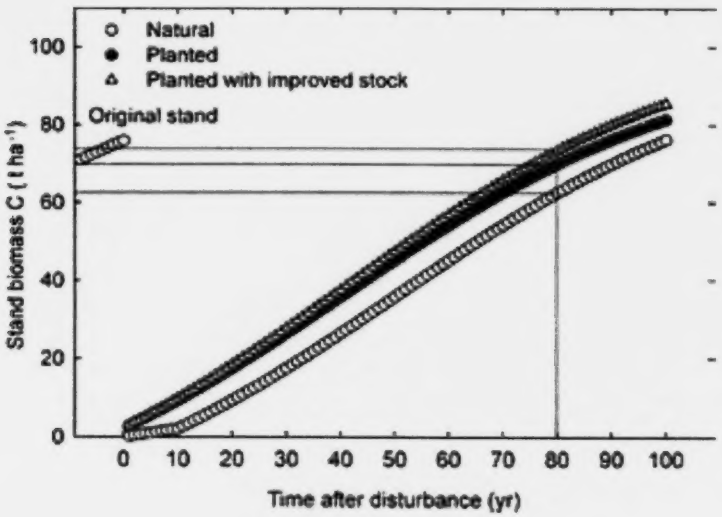

Figure 3.1.
Predicted black spruce total stand biomass C with natural regeneration, regeneration by planting non-improved stock, and regeneration by planting stock with 5% increased volume and biomass growth rates through tree improvement. Differences in biomass C at 80 years after disturbance are emphasized by straight lines.

Published provincial statistics on the planting of improved trees since 1990 are not available. However, one estimate is that about 20% of the seedlings planted in northeastern Ontario in 2002 were produced using improved seed (D. Joyce, 2004, OMNR, pers. comm.). For the purpose of these scenarios, it is assumed that a 5% increase in both growth rate and C sequestration is obtained from genetic improvement, and that use of improved seed increased at a constant rate from 0% in 1996 to 15% of provincial planting in 2000. The proportion of the planted area regenerated using improved seedlings is assumed to have reached 20% in 2001 and will increase by 5% per year after that. The area planted annually from 1997 to 2000 is based on OMNR statistics (OMNR 2001a,b,c). From 2001 onward, we assume an annual area planted equal to the mean area planted between 1990 and 2000.

The realized gains from genetic improvement depend on site factors and the sequence of silvicultural practices applied. For example, the gains in growth predicted for progeny trials in Ontario are based on plantations that were carefully tended after planting. Achieving the anticipated increases in growth in operational settings requires that genetically improved planting stock are of high quality and receive the benefits of good handling, planting, site preparation, tending, and thinning (Nienstaedt and Jeffers 1975).

Scenario 1. Carbon accumulation during the first Kyoto accounting period

Modelled total stand C accumulation rates for Site Class 2 black spruce stands, with and without genetic improvement, are shown in Figure 3.2. The difference in stand C for improved versus either non-improved or naturally regenerated stands was calculated from these curves for planting for the period from 1997 through 2009 (Table 3.1). Improved stock were planted on about 2,500 ha in 1997. By 2012, these plantations will have increased C storage by almost 1,700 t compared to a similar area planted with non-improved stock. The C gain in 2012 from planting improved stock in 1997 compared to natural regeneration of an equivalent area would be over 40,000 t (Table 3.1). Under the assumed increase in rate of planting with improved nursery stock from 1997 to 2009, about 76,000 t more C would be sequestered in 2012 from combined artificial regeneration and genetic improvement. Relative to naturally regenerated stands, the gain in C in 2012 from planting genetically improved stock is almost 2.2 Mt (Table 3.1).

However, under the Kyoto methodology for estimating the C gain, only the 2008 to 2012 gains would be credited (Lines Ai and Aii in Fig. 3.2). Estimated in this way, the gain compared to natural regeneration would be only about 1.9 t ha^{-1} on the just more than 2,500 ha planted in 1997, for a total C gain of about 4,700 t, compared to the total increase of almost 2.2 million t C. The gain in C from planting genetically improved stock compared to non-improved stock in 1997, accounted for using the Kyoto methodology, would be 0.16 t ha^{-1}, which over 2,500 ha provides added storage of just 400 t C. From the planting of genetically improved stock from 1997 to 2007, the C gain from 2008-2012 simulated using CBM-CFS2 would be about 23,000 t compared to non-improved stands and almost 630,000 t compared to natural regeneration.

Scenario 2. Carbon accumulation over an 80-year rotation

An average Site Class 2, naturally regenerated black spruce stand contains 88 m^3 gross merchantable volume at age 80 (Plonski 1971). Assuming the stand regenerated after clearcutting, total stand C at age 80 is estimated to be about 200 t ha^{-1} (Fig. 3.2). Planting clearcuts with non-improved stock results in total stand C at age 80 of 216 t ha^{-1} while a genetically improved stand with a 5% faster growth rate would contain about 219 t C ha^{-1} at age 80 (Fig. 3.2). Therefore, stands with a 5% genetic improvement in volume and biomass growth would contain about 3 t more C ha^{-1} at age 80 compared to stands of non-improved stock and about 19 t ha^{-1} more C compared to stands regenerating naturally. Using 2070 as a reference year, the additional C storage achieved by planting genetically improved stock from 1997 to 2012 would total about 1.1 Mt compared to non-improved stock and almost 8.7 Mt compared to natural regeneration (Table 3.2).

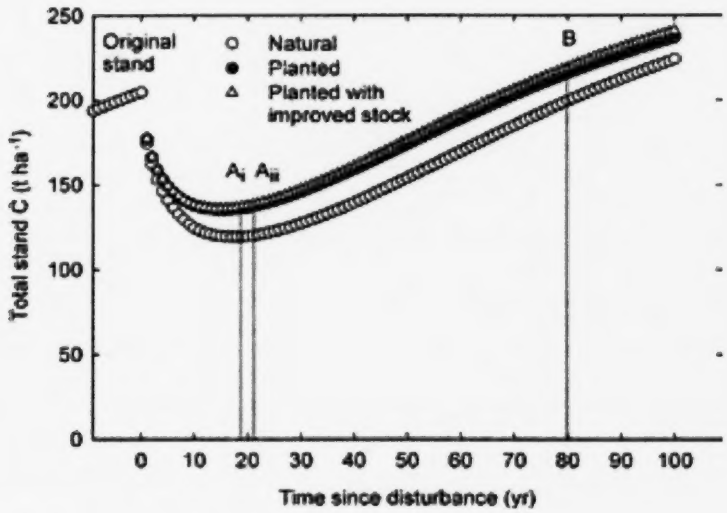

Figure 3.2. Predicted effects of planting genetically improved black spruce stock on total stand C. Vertical lines labeled 'Ai' and 'Aii" are the ages that stands naturally regenerated or planted with non-improved stock in 1990 would be in 2008 and 2012. Line B is drawn at age 80, a typical harvest age for naturally regenerated black spruce stands in Ontario.

Table 3.1. Increases in total stand C in 2012 due to a 5% gain in growth rates from planting genetically improved stock compared to planting non-improved stock or natural regeneration.

Year planted	Planted area receiving improved seedlings (%)	Area successfully planted with improved stock (ha)	C gain in 2012 vs. non-improved planted stock (t ha^{-1})	C gain in 2012 vs. natural regeneration (t ha^{-1})	C gain in 2012 vs. non-improved stock (t)	C gain in 2012 vs. natural regeneration (t)
1996	0	0	0.71	16.4	0	0
1997	3.8	2,528	0.66	16.0	1,681	40,460
1998	7.5	5,718	0.63	15.6	3,573	89,308
1999	11.3	9,454	0.59	15.2	5,530	143,525
2000	15.0	11,032	0.54	14.7	6,012	162,176
2001	20.0	13,121	0.50	14.1	6,626	185,559
2002	25.0	16,401	0.47	13.6	7,651	222,263
2003	30.0	19,682	0.43	12.5	8,424	246,867
2004	35.0	22,962	0.39	11.5	8,967	264,923
2005	40.0	26,242	0.35	10.5	9,263	275,884
2006	45.0	29,522	0.32	9.5	9,329	279,666
2007	50.0	32,803	0.28	8.4	9,168	276,117
2008	55.0	36,083		Negligible	Negligible	Negligible
2009	60.0	39,363		Negligible	Negligible	Negligible
Total		**266,586**			**76,226**	**2,186,749**

Table 3.2. Increases in total stand C in 2070 due to a 5% gain in growth rates from planting genetically improved stock compared to planting non-improved stock or natural regeneration from 1997 to 2012. Total C gain values assume no harvest occurs prior to 2070.

Year planted	Planted area receiving improved stock (%)	Area successfully planted with improved stock (ha)	C gain in 2070 vs. non-improved planted stock (t ha^{-1})	C gain in 2070 vs. natural regeneration (t ha^{-1})	C gain in 2070 vs. non-improved stock (t)	C gain in 2070vs. natural regeneration (t)
1997	3.8	2,528	3.3	20.7	8,202	52,231
1998	7.5	5,718	3.2	20.8	18,325	118,301
1999	10.5	9,454	3.2	21.0	29,921	196,787
2000	15.0	11,032	3.1	21.1	34,529	232,161
2001	20.0	13,121	3.1	21.2	40,544	276,436
2002	25.0	16,401	3.1	21.3	50,024	347,545
2003	30.0	19,682	3.0	21.3	59,242	419,416
2004	35.0	22,962	3.0	21.4	68,082	489,617
2005	40.0	26,242	2.9	21.6	76,758	562,607
2006	45.0	29,522	2.9	21.6	85,172	636,327
2007	50.0	32,803	2.8	21.7	93,160	707,293
2008	55.0	36,083	2.8	21.8	101,032	781,991
2009	60.0	39,363	2.8	21.8	108,446	857,136
2010	65.0	42,644	2.7	21.9	115,564	928,606
2011	70.0	45,924	2.7	21.9	122,617	1,004,722
2012	75.0	49,204	2.6		129,161	1,076,291
Total		**402,682**				**8,687,467**

Scenario 3. Carbon accumulation under different rotation ages

The faster growth of genetically improved stands decreases the length of the rotation needed to provide a given volume of timber. At 80 years, a typical rotation length for naturally regenerated black spruce stands on Class 2 sites, total biomass C averages about 63 t ha^{-1}. Equivalent amounts of biomass C would be present in planted, non-improved and improved stands, at ages 70 and 67 years, respectively, after establishment (Fig. 3.3). The shorter technical rotation in planted stands reduces total forest C for up to 13 years relative to that in non-harvested naturally regenerated stands. As a result of the removal and processing of biomass C into forest products, the decay of logging debris and other litter, and the low rates of biomass C accumulation in young stands, naturally regenerated, planted, and genetically improved stands contain 200, 137, and 134 t C ha^{-1}, respectively, 80 years after establishment. These three plantation types do not contain similar total stand C until 86 years after establishment (i.e., 6 years after harvest of the naturally regenerated stand) (Fig. 3.3).

Where genetically improved stands planted from 1990 to 2020 were harvested early, the increased storage of C in 2070 would total almost 17 Mt relative to naturally regenerated stands (Table 3.3). This increase occurs despite the fact that stands planted with improved stock from 1997 to 2002 would contain less C in 2070 than non-harvested naturally regenerated stands. The lower C occurs in stands harvested earlier due to the reduced size of the DOM and biomass pools for a number of years after harvest, as described above. Considering all genetically improved stands planted between 1997 to 2020, C storage in 2070 would total almost 17 Mt more C than had the stands been allowed to regenerate naturally (Table 3.3).

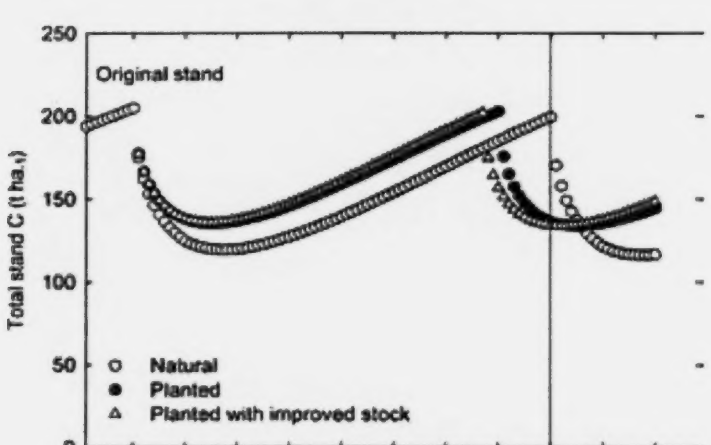

Figure 3.3.
Total stand C in three forest stands growing at different rates and harvested when reaching total tree biomass C of 63 t ha^{-1}. Naturally regenerated stands are harvested at age 80 (vertical line), stands planted with non-improved stock are harvested at age 70, and genetically improved stands are harvested at age 67.

Table 3.3. Increases (losses) in stand C in 2070 due to a 5% gain in growth rates from planting genetically improved stock compared to natural regeneration from 1996 to 2020. Total C gains assume harvesting genetically improved stands at 67 years and naturally regenerated stands at 80 years.

Years since stand establishment	Initial planting year	Age of improved stands (yr)	Area of improved stands (ha)	C in improved stands in 2070(t ha^{-1})	C in naturally regenerated stands in 2070(t ha^{-1})	Gain (loss) of C (t)
74	1996	7	0	141.3	191.1	0
73	1997	6	2,528	143.9	189.6	(115,522)
72	1998	5	5,718	147.2	188.2	(234,320)
71	1999	4	9,454	151.4	186.7	(333,188)
70	2000	3	11,032	156.9	185.1	(311,128)
69	2001	2	13,121	164.4	183.7	(253,815)
68	2002	1	16,401	175.0	182.2	(118,193)
67	2003	0	19,682	201.9	180.6	419,416
66	2004	66	22,962	200.4	179.1	489,617
65	2005	65	26,242	199.0	177.6	562,607
64	2006	64	29,522	197.6	176.0	636,327
63	2007	63	32,803	196.0	174.5	707,293
62	2008	62	36,083	194.6	172.9	781,991
61	2009	61	39,363	193.1	171.3	857,136
60	2010	60	42,644	191.5	169.8	928,606
59	2011	59	45,924	190.1	168.2	1,004,722
58	2012	58	49,204	188.6	166.7	1,076,291
57	2013	57	52,484	187.0	165.1	1,152,872
56	2014	56	55,765	185.5	163.5	1,224,313
55	2015	55	59,045	184.0	161.9	1,301,410
54	2016	54	62,325	182.4	160.4	1,372,588
53	2017	53	65,606	180.9	158.9	1,443,452
52	2018	52	65,606	179.3	157.3	1,448,374
51	2019	51	65,606	177.7	155.8	1,439,581
50	2020	50	65,606	176.2	154.2	1,444,043
					Net gain (loss) of C in 2070	**16,924,473**

3.2 Stand Density Management

Stand density management, or thinning, refers to partial harvesting in immature stands and is used primarily to accelerate diameter growth and improve the average form of the remaining trees. Periodic reductions in stand density in managed forests seek to simulate natural processes, accelerate stand development, balance stand with tree productivity, and avoid yield losses to mortality (Smith et al. 1997). Suppressed crop trees are harvested in anticipation of their eventual mortality, to reduce intraspecific competition, and temporarily increase light, soil moisture, and nutrient availability. This reduction in stand density supports crown expansion and increased stem diameter and volume growth of the residual trees (Waring et al. 1981, Smith et al. 1997, Aussenac 2000). The improved conditions for growth and, in some cases, increased physiological activity, may increase production efficiency (i.e., growth per unit leaf area) of trees in thinned stands (e.g., Table 3.4) (Waring et al. 1981, Binkley and Reid 1984, Mitchell et al. 1996).

The timing and intensity of density regulation can be determined by the interrelationship between the number of trees per unit area, mean tree size (e.g., diameter and height), and stand volume or basal area per unit land area for a given species (Reineke 1933, Gingrich 1967, Drew and Flewelling 1979). Residual density is generally selected to create the best combination of tree and stand productivity depending on species, site, and management objective (e.g., pulpwood, sawlogs). Optimum stand density typically occurs at intermediate levels (Crow et al. 1981, Johnstone 1997, Miller 1997). At low densities, individual tree growth and size are maximized but stand productivity is well below its potential. At comparatively high stand densities, gross stand production is greatest, but inter- and intraspecific competition reduces individual tree growth rates and average stem size. Competition-induced mortality also increases at higher densities, further reducing net stand production and yield.

Stand density management to improve stand growth and yield was not practised in Canada on a large scale until the 1990s, when depletion of natural forests and withdrawals of Crown land from timber production led to concerns about future wood supply. While some forest companies in eastern Canada now obtain as much as 40% of their wood supply from thinning, only about an average of 14,000 ha yr^{-1} have received some form of density regulation in Ontario since 1990 (CCFM 1997, Day and Nanang 1997). In this section, we discuss three general types of stand density management: pre-commercial thinning, crop tree release (or crown thinning), and commercial thinning and their influence on stand C balance.

Table 3.4. Stand characteristics of 36-year-old, naturally established Douglas-fir (*Pseudotsuga menziesii* Franco.) stands grown for 15 years at five average stand densities (number of trees ha^{-1})*. Values are means followed by standard error (in parentheses). (Adapted from Waring et al. 1981.)

Feature	Stand density (trees ha^{-1})				
	173	304	477	593	1977
DBH (cm)	35.9 (0.8)	34.2 (1.0)	28.9 (1.2)	29.8 (0.8)	18.3 (1.2)
Height (m)	25.5 (0.3)	25.6 (0.3)	25.5 (0.4)	25.5 (0.3)	24.0 (0.5)
Basal area (m^2 ha^{-1})	17.5 (0.8)	27.9 (1.6)	31.3 (2.6)	41.3 (2.2)	52.5 (6.9)
Stand volume (m^3 ha^{-1})	224.7 (13.2)	348.1 (19.7)	352.8 (38.0)	475.0 (33.1)	451.3 (76.8)
Total tree C (Mg ha^{-1})**	79.4	122.9	124.6	167.7	159.4
Tree volume increment (m^3 yr^{-1})	0.124 (0.003)	0.092 (0.005)	0.059 (0.007)	0.049 (0.005)	0.010 (0.003)
Stand leaf area index (m^2 m^{-2})	3.6 (0.2)	6.0 (0.3)	7.2 (0.5)	8.6 (0.4)	12.0 (1.8)
Stem wood production efficiency (g m^{-2} leaf area)	235 (5)	185 (2)	157 (6)	135 (7)	66 (3)

* Stand density was maintained by five successive thinning treatments over a 12-year period.
** Estimated by the method of Birdsey (1992).

Pre-commercial thinning

Pre-commercial thinning, or 'juvenile spacing' refers to stand density management that is usually aimed at improving crop tree spacing but in which the trees removed by thinning have no commercial timber value. Thinning to obtain a wider, more uniform spacing of young, overly dense naturally regenerated stands is performed to reduce intraspecific competition, prevent growth stagnation, minimize growth losses to mortality, and improve the growth and quality of selected crop trees during an early phase of stand development. Pre-commercial thinning of overly dense stands can significantly improve growth and yield of many Canadian forest tree species by increasing the rate of C accumulation and decreasing the technical rotation (Farnum et al. 1980, Haavisto et al. 1991, Johnstone 1997, Rice et al. 2001). The need for precommercial thinning commonly arises in naturally regenerated stands of black spruce, white spruce, balsam fir, white birch, and *Populus* sp. but is most likely in jack pine and lodgepole pine (Steneker 1974, Labonte and Nash 1978, Piene 1978, Pothier and Margolis 1991, Burns et al. 1996, Johnstone 1997, Brisette et al. 1999, Rice et al. 2001).

Overstocked naturally regenerated jack pine stands are common in northwestern Ontario, developing after wildfire or aerial and natural seeding, with densities of 50,000 to 100,000 stems ha^{-1} sometimes occurring (White and Waldram 1991). The influence of pre-commercial thinning of jack pine on C storage in forest biomass and in wood products is illustrated by the results of Barbour et al. (1994), presented in Table 3.5. Closer initial spacing (i.e., 1.7 m on average) eventually reduced mean stem diameter, but increased total merchantable volume production and total tree C due to the retention of more trees per hectare. However, wider (3.4 m) spacing decreased the rotation for stems $\geq$ 16 cm butt diameter by 25%. Further, 13% and 28% of stems had a butt diameter $\geq$ 25 cm in the 2.6 m and 3.4 m spacing, respectively, as compared with no trees of this size in the control treatment. Wider spacing slightly and negatively affected wood properties, with stem wood density being 4 to 7% lower than the unspaced stand, which may influence the type of wood product that can be produced. When interpreted in terms of C storage, the net gain in stem wood mass and tree C at closer spacing likely outweighs any slight reduction in structural properties of the stem wood.

Table 3.5. Stand and wood attributes of a 65-year-old jack pine stand 38 years after thinning to two average spacings (2.6 m, 3.4 m) as compared with an unthinned control (1.7 m spacing). (Adapted from Barbour et al. 1994.)

Stand and stem wood features	Initial tree spacing		
	1.7 m	2.6 m	3.4 m
(a) Stand			
No. stems ha^{-1}	1372	1038	717
Average DBH (cm)	15.5	16.5	18.8
Average height (m)	17.1	16.6	17.6
Basal area (m^2 ha^{-1})	26.6	23.2	20.4
Total volume (m^3 ha^{-1})	201.6	170.9	159.8
Merchantable volume (m^3 ha^{-1})	181.0	155.0	147.3
Tree C (Mg ha^{-1})*	71.2	60.4	56.4
(b) Wood and forest products			
No. stems > 16 cm diameter	1166	913	703
No. stems > 25 cm diameter	0	136	198
Straight stems (%)	<10	~33	~20
Stem wood mass (Mg ha^{-1})	81.3	66.3	61.9
Relative density	0.456	0.433	0.421
Ring width (mm)	0.718	0.913	1.017

* Estimated by the method of Birdsey (1992).

Crop tree release of hardwoods

Dense stands of northern hardwood species that often result after clearcutting require pre-commercial thinning or crop tree release to accelerate stand development (Heitzman and Nyland 1991, Miller 2000). In young, even aged sapling and pole-sized stands where trees of more desirable species and/or higher quality are easily identified, adjacent trees with dominant and codominant crown class are removed to favour the growth of selected crop trees. This crown thinning improves the diameter and volume growth of crop trees by removing competition and increasing crown growing space (Heitzman and Nyland 1991, Miller 2000). Crop tree release has relatively little effect on height growth, but significant increases in stem volume growth (~ 30 to 120%) have been reported for many commercially important hardwood species of Ontario (Heitzman and Nyland 1991, Woods et al. 1996, Miller 2000). This growth response is generally positively correlated with the degree of crown release applied (Woods et al. 1996, Ward 2002). The large diameter growth responses to crop tree release considerably reduce the technical rotation (Steneker 1974, Erdemann et al. 1975, Jones et al. 1989). Crop tree release to increase crown growing space has also significantly increased diameter and volume when applied to older, sawtimber-sized northern hardwood species (Smith et al. 1994, Ward 2002).

The degree of release applied must achieve a balance between increased resources and growth, while maintaining stem wood quality. Where crop tree release creates canopy openings that are too large, high light availability may stimulate the development of dormant buds along tree stems giving rise to epicormic branches. These branches may persist and reduce the length of clear wood formed along the bole. Epicormic branching, the formation of stem forks, and delayed natural pruning after release all increase the number of limb-related defects, a major cause of degraded log quality in hardwoods. This loss of stem quality is greatest with heavier release treatments, lower crown classes, and in more shade-tolerant species, but can be reduced by investment in pruning.

Commercial thinning

Thinning of many of Ontario's commercially important conifer species can significantly increase volume increment and yield and reduce the technical rotation for producing larger logs (e.g., poles and sawlogs) while providing an additional mid-rotation source of merchantable fibre (Bella and DeFrancheschi 1974, Stiell 1980, Stiell et al. 1994, Burgess and Robinson 1998, Woods and Penner 2000). For example, thinning to remove hardwoods from a white pine mixedwood stand increased stem diameter and volume increment and reduced the rotation for pine sawlogs by 20 to 30 years (Stiell 1984, Stiell et al. 1994). Significant stem growth increments and reductions in the technical rotation of northern hardwood species by thinning have also been reported (Marquis 1969, Roberge 1975, Erdemann et al. 1985, Perala 1991). Basal area and diameter growth rates of heavily thinned northern hardwood stands may increase by >100% within 5 to 10 years of thinning (Ellis 1979, Erdemann et al. 1985, Lorimer 1997). As a result, the time required for dbh to increase from 30 to 50 cm in sugar maple, white ash, and basswood was projected to decrease by as much as 20 to 30 years by thinning at 10-year intervals (Singer and Lorimer 1997). Growth improvement by regular thinning of a pole-size red maple stand in northern Michigan was estimated to reduce the technical rotation for sawlogs from 127 to 85 years (Erdemann et al. 1985).

Density regulation and stand carbon balance

Although tree size varies with stand density, stand volume is relatively constant at any age over a range of densities (Smith et al. 1997). Thinning increases the growth of individual trees, but at the expense of total stand production and C storage (Krankina and Harmon 1994). Utilizing biomass that would otherwise be lost to mortality can significantly increase merchantable volume and yield (Day and Nanang 1997). However, when total biomass in mortality and thinnings are accounted for, the gross production of thinned and unthinned stands is generally comparable (von Althen and Stiell 1990, Schroeder 1992, Krankina and Harmon 1994).

Therefore, despite improving the growth of the residual trees, thinning neither increases site productivity nor the amount of C sequestered per unit land area. As well, removal of smaller suppressed trees that would eventually die in unmanaged stands can lower C storage in the DOM pool (Krankina and Harmon 1994). Rather, thinning concentrates growth and biomass accumulation on a smaller number of selected crop trees. However, by concentrating growth on fewer, faster-growing trees, thinning can reduce the time needed to grow stems of a particular size, i.e., the technical rotation for a desired forest product. For example, enhanced stem wood increments in thinned conifer stands can decrease rotations for sawlogs by 20 to 30 years (Stiell 1980, Burgess and Robinson 1998).

Thinning can also influence the amount of C stored in forest products. If harvested biomass is incorporated into wood products that have a slower decay rate than DOM from individual tree mortality, the net C balance of the stand can be improved. If thinning increases the yield of solid wood products with a relatively long storage life, is substituted for more energy intensive structural materials, or if thinned material is burned in place of fossil fuels, the net C balance of thinned stands increases further (Micales and Skog 1997, Skog and Nicholson 1998).

Thinning has little long-term effect on soil C levels. Thinning removes aboveground tree biomass, while root systems from harvested trees remain on site and are incorporated into the soil. A transient change occurs in DOM and soil C after thinning that depends on the relationship between the rate of litter production, the amount of harvesting residues left on site, and DOM decomposition rate. Increased decomposition due to elevated forest floor temperatures in more open stands combined with reduced litter fall after thinning may briefly change the DOM pool from a C sink to a source (Piene 1978, Piene and Van Cleve 1978, Thibodeau et al. 2000). This shift in DOM C balance is short-lived (< 5-10 years) as lateral crown expansion and understory development shelter and reduce forest floor temperatures and litter inputs return to pre-thinning levels (Peterson et al. 1997). For this reason, the relative amounts that decomposition and DOM C are changed by regular thinning is much less than after clearcutting (Thornley and Cannell 2000). Although decreased DOM C following stand density reduction has been reported, significant reductions in this C pool relative to undisturbed stands are usually only found with heavier thinning (Wollum and Schubert 1975, Vesterdahl et al. 1995, Rollinger et al. 1997).

Thinning can have a positive indirect effect on C storage by reducing stand susceptibility to snow and ice damage, removing weak trees susceptible to insects or disease, increasing the vigour and pest resistance of residual trees, and lowering the risk of wildfire, hence reducing forest C losses to these natural disturbances (Mitchell et al. 1983, Janas and Brand 1988, Safranyik et al. 1998, Graham et al. 1999, Parker et al. 2000). Thinning of mid-tolerant species such as white pine and white spruce can also be used to cultivate advance regeneration, thus reducing the loss of DOM C and decreasing the rotation of the future stand relative to clearcutting (Burgess and Robinson 1998). The potential negative effects of thinning on stand C balance (e.g., windthrow, logging damage, soil compaction, and increased susceptibility to some insects and diseases) can be minimized by carefully planning partial harvesting operations (see Section 5.1) (Ruel 1995, Hayes et al. 1997, Safranyik et al. 1998).

3.3 Fertilization

Stand growth and productivity are closely associated with the availability of macro- and micronutrients. In boreal and temperate forests, nitrogen (N) deficiency is widespread and common (Binkley 1986, Vitousek and Howarth 1991, Johnson 1992b) and reduces stand productivity by decreasing leaf area, light interception, and to a lesser extent photosynthesis (Albrekston et al. 1977, Binkley 1986, Reich et al. 1997). Since stand leaf area index (LAI) is strongly related to NPP, low N availability can substantially decrease the growth and C sequestration of northern forests (Reich et al. 1997). As a result, N is the element most often applied in forest fertilization and generally increases stem diameter and volume growth in both conifer and hardwood stands (White and Leaf 1958, Baule and Fricker 1972, Krause et al. 1987). The response to N addition is often improved when other nutrients are supplied as well to alleviate any fertilization-induced nutrient deficiencies or imbalances (Brockley 1990, Fisher and Binkley 2000). Fertilization with N in combination with phosphorus (P) and potassium (K) has also been shown to increase tree growth and stand NPP by increasing leaf area, photosynthesis, and/or production efficiency (Brix 1981, 1983, Binkley and Reid 1984, Linder and Rook 1984, Gower et al. 1992, Mitchell et al. 1996). Although less common, forest growth limitations caused by deficiencies in P, K, Mg (magnesium), calcium (Ca), S (sulfur), and other elements, and its alleviation by fertilization, has also been reported (Stone 1953, Leaf 1968, Gagnon et al. 1976, Binkley 1986, Brockley 1990). Fertilization with these other mineral elements has also been used successfully to correct nutrient imbalances caused by atmospheric N deposition, base-cation leaching due to acid precipitation, and past land use (Leaf 1968, Fyles et al. 1994, Long et al. 1997).

Variation in response to fertilization exhibited among and within species is the result of the strong influence of site and stand (density, species composition, tree size and vigour) conditions on whether trees will benefit from nutrient amendments (Chappell et al. 1991, Fisher and Binkley 2000). Maximum fertilization response is attained when nutrient additions are balanced with the nutrient demand of forest stands relative to site nutrient availability, a relationship that varies with stage of stand development (Miller 1981, Oliver and Larson 1990, Johnson 1992b). Fertilization response is typically greater on poorer quality sites with low nutrient reserves or specific elemental deficiencies, provided other resources such as soil moisture are sufficient (Mitchell and Chandler 1939, Foster and Morrison 1983, Krause et al. 1987, Allen 1987). Fertilization response is also affected by patterns of immobilization and mineralization of the nutrients after they are applied.

Stand density has a strong influence on fertilization response, with stands of moderate stocking providing the growing space needed for increased leaf area production, crown expansion, and improved photosynthetic potential following nutrient amendment (Madgwick et al. 1970, Miller 1981, Allen 1987). Overstocked stands may respond poorly to fertilization due to accelerated self-thinning and increased mortality of trees in lower crown positions, which decreases the net merchantable volume gains (Allen 1987, Weetman et al. 1987). In some cases, thinning prior to fertilization improves the response by allowing space for crown expansion and increasing the resources (i.e., light, soil moisture) needed for residual trees to take full advantage of increased nutrients (Kawana et al. 1969, Weetman et al. 1980, Mitchell et al. 1996, Yang 1998). While the growth of individual trees may be improved by this treatment combination, total stand biomass production is decreased for a period, due to lower overstory stand density after thinning (Stegemoeller and Campbell 1991, Valinger 1993, Yang 1998). In time, the total volume or biomass of thinned and fertilized stands may recover to that of untreated stands due to increased growth increment of individual trees. Significant additive effects of thinning on the growth response to N fertilization are generally not exhibited, but have been reported for coastal Douglas-fir (Mitchell et al. 1996). Thinning jack pine and black spruce stands in eastern Canada prior to fertilization did not improve total stand growth (Weetman et al. 1980, Morrison and Foster 1990), although individual tree size increased.

Ecological sustainability of forest fertilization

Forest fertilization is rarely used operationally in Canada despite the large growth responses exhibited. From 1990 to 2003, about 121,000 ha of forests were fertilized in Canada, or about 8,500 ha yr^{-1}, almost all of which was provincial Crown forest land in British Columbia (CCFM 2005). This practice is currently prohibited or discouraged by various forest certification protocols due to uncertainty about its potential adverse effects on environmental quality and ecological sustainability (Bowen and Nambiar 1984, Fox 2000). Forest fertilization is not currently approved for operational use in Ontario under the conditions of the Timber Class Environmental Assessment (OEAB 1994). Of primary concern are the effects of fertilizers on water quality and aquatic ecosystems. However, much scientific research suggests that proper fertilization with N and P at operational rates has not affected the quality of water draining from forested ecosystems and has no significant adverse effects on species composition and productivity of aquatic ecosystems (Allen 1987, Binkley et al. 1999, Fisher and Binkley 2000).

A second concern with fertilization is that it may alter the composition, structure, and diversity of understory plant species (Prescott et al. 1995, Fahey et al. 1998, Thomas et al. 1999, Turkington et al. 2002). The abundance of ericaceous shrubs, bryophytes, and lichens typically declines after N fertilization, while grasses increase in abundance (Miller and Miller 1976, Kellner 1993, Prescott et al. 1995, Turkington et al. 2002). The limited research conducted to assess the influence of fertilization on understory plant diversity shows variable responses (Kellner 1993, Kellner and Marshner 1991, Turkington et al. 2002), but the negative effects reported for certain species and functional groups are cause for concern (Thomas et al. 1999).

Changes in understory species may affect wildlife habitat quality by altering the vertical structure of understory vegetation and the nutrient content and abundance of browse species. Fertilization may increase the nutritional quality and palatability of woody and herbaceous understory species and increase browsing by wildlife (Ward and Bowersox 1970, Turkington et al. 2002). However, where excessive browsing hinders the regeneration of crop tree species, fertilization can be used to increase height growth rates, which decreases the period of vulnerability of crop trees to herbivory, as well as to enhance the growth of non-crop browse species (Auchmoody 1982, Marquis et al. 1992). Fertilization has also been reported to increase flower and seed production of many tree species, an important food source for birds and small mammals (Mergen and Voigt 1962, McLemore 1975, Valinger 1993, Long et al. 1997).

Forest fertilization and stand carbon balance

Most silvicultural treatments redistribute C among individuals, improve the value of wood produced, or decrease the technical rotation for a given product, rather than increase the C storage potential of stands. By comparison, fertilization is one of the few techniques that can be used to consistently improve growth and forest C storage (Allen 1987, Schroeder 1991, Hoen and Solbrig 1994, Fisher and Binkley 2000) by temporarily improving site nutrient conditions and increasing NPP (Miller and Miller 1976, Wittwer et al. 1975, Mitchell et al. 1996, Yang 1998). For example, single applications of urea fertilizer to fully stocked, second-growth Douglas-fir stands in the Pacific Northwestern U.S. increased stem volume growth by 2 to 4 m^3 ha^{-1} (9 to 23%) annually for 8 to 15 years, roughly equivalent to an increase in aboveground biomass of 1.3 to 2.7 t ha^{-1} yr^{-1} (Chappell et al. 1991). Aboveground NPP of a 50-year-old Douglas-fir stand was increased by 4.1 t ha^{-1} yr^{-1} for two years after a single N application (Gower et al. 1992). Urea fertilization of a 45-year-old jack pine stand resulted in a net increase in total biomass production of 18.4 t ha^{-1} within three years, roughly 73% of which was stem wood (Morrison and Foster 1977). Fertilizer response varies with the amount, frequency and timing of fertilizer application (Allen 1987, Fisher and Binkley 2000). Long-term, repeated fertilization, particularly when used to achieve optimal nutrition, often results in even larger, more prolonged C gains (Weetman et al. 1995, Kishchuk et al. 2002).

Forest fertilization on N-deficient sites consistently increases tree growth, although vegetation typically recovers only a small portion of the applied N (Foster and Morrison 1983, Johnson 1992b, Bell et al. 1997). Nitrogen retention in forests is linearly related to the amount applied, and averages 25% (range 6 to 65%) and 35% (range 15 to 67%) in vegetation and soil, respectively, with the remainder lost to leaching and volatilization (Chappell et al. 1991, Johnson 1992b, Fisher and Binkley 2000). The benefits of fertilization to tree growth may be reduced by the fairly rapid incorporation of applied N into litter and soil through uptake by heterotrophic organisms or non-biological, chemical reactions between N and soil organic matter (Johnson 1992b, Fisher and Binkley 2000). Release of this immobilized N depends on the rate of N mineralization, which occurs relatively slowly in northern forests. Since fertilization typically only increases site nutrient availability for 1 to 2 years, current annual increment of fertilized trees often returns to its pre-treatment rate within 5 to 10 years, although positive growth response of more than 20 years have been reported (Binkley and Reid 1984, Shepard and Mitchell 1990). Therefore, fertilization commonly does not permanently affect site productivity, and sustained yield improvement requires repeated applications (Cole 1995). Rather, it accelerates stand development and decreases the time needed to produce a given stem size (Miller 1981).

Although growth responses to a single fertilization generally last less than 10 years, fertilization may increase long-term productivity of degraded sites where nutrient inputs are large relative to those available in soil. For example, nutrients applied to pine plantations established on impoverished agricultural land tend to become a permanent component of site nutrient capital (Stone and Kszystniak 1977, Fox 2000). This long-term gain in productivity is caused in large part by nutrient retention in the greater amounts of soil organic matter accumulated in fertilized stands (Grigal and Vance 2000). This suggests that fertilization may improve afforestation of marginal agricultural land to increase the area of forests and forest C storage. Conversely, repeated fertilization may be necessary to maintain site productivity where short-rotation, intensive plantations of fast-growing species are established and a large proportion of site nutrient capital is removed in the harvested fibre.

Nutrient amendments increase forest C storage in total biomass, litter, and mineral soil by increasing GPP and NPP. The relative increase in above- and belowground biomass C after fertilization varies with species and site conditions. Fertilization commonly increases stem growth and wood production by reducing carbon allocation to roots (Farrell and Leaf 1974, Linder and Rook 1984, Gower et al. 1992, Haynes and Gower 1995, Fahey et al. 1998) but increased belowground biomass production after fertilization has been reported (Safford 1974, Harding and Jokela 1994, Sword et al. 1996, Fox 2000). In most cases, fertilization has a significant positive effect on C storage in the forest floor and mineral soil (Nohrstedt et al. 1989, Shepard and Mitchell 1990, Harding and Jokela 1994). A review of fertilization trials revealed that N application increases mineral soil C by about 20% (Johnson and Curtis 2001). This is likely due to increased litterfall coupled with little change or a reduction in decomposition rate associated with fertilization (Miller and Miller 1976, Prescott 1995, Haynes and Gower 1995, Mitchell et al. 1996, Currie et al. 2003).

Fertilization may indirectly affect C storage by causing nutrient imbalances that alter tree resistance to forest insects and pathogens (Foster 1968, Baule and Fricker 1972, Smirnoff and Bernier 1973, Fisher and Binkley 2000). Nitrogen fertilization of lodgepole pine increased tree vigour and resistance to mortality by bark beetles (Waring and Pitman 1985). On the other hand, 10-year-old white pine plantations fertilized with K suffered a higher incidence of white pine weevil (*Pissoides strobi* Peck.) damage, perhaps due to increased height growth and terminal shoot diameter (Xydias and Leaf 1964). However, pest-related effects on growth associated with fertilization are typically small relative to the biomass gains consistently obtained from this practice (Fisher and Binkley 2000).

Forest fertilization to increase carbon storage in Ontario's forests

Fertilization research in Canada began early in the 1900s and focused on seedling nutrition in forest nurseries and nutrient application at the time of planting to improve plantation establishment (Armson 1967, Rennie 1972, Foster and Morrison 1983). Major efforts to examine the potential value of fertilization to increase wood production did not occur until the late 1960s when several national and provincial programs were established (Foster and Morrison 1983). The primary objective of these initiatives was to evaluate the potential of fertilization to increase fibre production in anticipation of projected shortfalls in wood supply for the pulp and paper industry. Trials emphasized fertilization of natural, fully to over-stocked, pole-sized to semi-mature single species or mixedwood stands of major pulp species that were within 10 years of harvest. The 10-year results of these programs indicate that growth of jack pine stands consistently increased after N fertilization (Table 3.6). Growth responses of black, white, and red spruce (*Picea rubens* Sarg.), and balsam fir stands were often positive, but generally lower than those of jack pine (Foster and Morrison 1983, Krause et al. 1987, Weetman et al. 1987).

Compared with boreal and temperate conifers, most northern hardwood species have been less responsive to fertilization, but poor site selection (i.e., sites were not nutrient limited) for testing may limit the relevance of these early fertilization trials (Ellis 1979, Lea et al. 1979, Safford 1973, Stankurt et al. 1989). Supplemental application of lime is sometimes required for positive growth response to NPK in northern hardwoods (Ward and Bowersox 1970, Safford 1973, Safford and Czapowskyj 1986, Wilmot et al. 1996). In eastern North American forests with acid soils and base cation deficiency, the application of lime, alone, or in combination with K, Mg, and Ca fertilizers improved vigour and growth of sugar maple (Fyles et al. 1994, Wilmot et al. 1996, Long et al. 1997). Trembling aspen responds rapidly to fertilization. Annual volume increment of aspen stands fertilized with NPK increased by 0.4 to 7.3 m^3 ha^{-1} over five years (Weetman et al. 1987) and up to 6.8 m^3 ha^{-1} after 15 years (Yang 1991). Repeated N fertilization of aspen stands in Alaska increased annual aboveground biomass increment 0.6 to 1.77 t ha^{-1} over nine years (Van Cleve and Oliver 1982).

Of Ontario's major commercial species for which information is available, jack pine is the best candidate for fertilization to increase C storage. Based on an average stem volume gain of 1.4 m^3 ha^{-1} yr^{-1} for jack pine in northern Ontario, a single application of N fertilizer to semi-mature, fully stocked jack pine stands would store an additional 4.9 t C ha^{-1} (total tree C) over 10 years. However, inorganic N fertilizer production is an energy-intensive process that generates substantial amounts of CO_2 and other GHGs (Beaton 1974, Schroeder 1991, Huettl and Zoettl 1992, Hoen and Solberg 1994). As well, N fertilization may alter fluxes of N_2O and CH_4 from forest soils, processes that must be accounted for to quantify the net C storage benefits of fertilization (Steudler et al. 1989, Castro et al. 1994, Hoover 2003). For example, emissions of CO_2 and NO_x during inorganic N fertilizer manufacture were estimated to offset 2 to 5% of the average potential C storage gain anticipated when N fertilizers are applied to conifer stands (Schroeder 1991, Hoen and Solberg 1994). For each hectare fertilized with a typical application rate of ammonium nitrate fertilizer (225 kg N ha^{-1}), approximately 1 t of CO_2 (and 270 kg C) is released to the atmosphere during manufacture. Volatilization of N fertilizer to N_2O could result in an additional release of 844 kg CO_2 (230 kg C)[1] . Using our jack pine example, this 500 kg C ha^{-1} would be offset by 10.2% of the total tree C gained over 10 years by fertilization. This scenario does not account for CO_2 emissions associated with transporting and applying this fertilizer. More attractive net C balances might be achieved through use of organic fertilizers from various sources (e.g., wastewater, livestock manure, municipal bio-solids) but these alternative sources will have environmental costs as well (Fisher and Binkley 2000).

The average aboveground biomass gain from a single mid-rotation fertilization of pure jack pine and black spruce stands is 22.6 and 9.0 m^3 ha^{-1}, respectively (Table 3.6). Based on this response rate, fertilizing 10,000 ha

[1]Assumes 0.0125 kg N_2O released per kg N^{-1} in fertilizer used in agricultural applications (Brentrup et al. 2001) and N_2O has CO_2 equivalent value of 300. The C offset value of 10.2% is presented to illustrate the hidden environmental costs associated with fertilization and other forest practices that need to be accounted for when considering options for increasing forest C storage.

of jack pine stands would increase stand biomass by 226,000 m^3 after 10 years, or about 56,500 t C. Assuming an equal annual growth increment over a 10-year period after fertilization, the increase in C in 10,000 ha of fertilized jack pine forest in the 5-year Kyoto accounting period between 2008 and 2012 would be about 28,250 t. If 10,000 ha of jack pine were fertilized each year from 2005 through 2011, the gain in C between 2008 and 2012 would total 163,850 t.

Table 3.6. Range and average values for 10-year total aboveground stand volume growth of major northern conifer species and forest types compared to unfertilized control treatments following a single mid-rotation fertilization.

Forest Type	Total volume gain (m^3 ha^{-1})	Fertilizer	Reference
Jack pine	3.2 – 11.6	NPK	Weetman et al. 1987
	5.0 – 22.9	N	Morrison et al. 1977
	4.6 – 8.8	N	Morrison et al. 1977
	4.5 – 21.7	NPK	Morrison et al. 1981
	-10.4 – 35.2	N**	Weetman and Fournier 1984a
	7.8 – 32.1	NPK	Weetman and Fournier 1984b
Average	**22.6**		
Black spruce	4.7 - 8.4	NPK	Weetman et al. 1987
	5.6 – 15.0	N	Weetman et al. 1980
	-5.0 – 4.3	NPK	Morrison et al. 1976
	5.0 – 25.2	NPK***	van Nostrand 1976
	-8.0 – 34.6	NPK	Morrison 1991
Average	**9.0**		
Black spruce – poplar	-7.4 – 16.5	NPK	Morrison 1991
Average	**4.6**		
Jack pine-black spruce	9.6 – 18.6	NPK	Weetman et al. 1987
Average	**14.1**		
White spruce	4.5 – 12.1	NPK	Weetman et al. 1987
	-2.2 – 27.8	NK	Gagnon et al. 1976
Average	**10.6**		
Red spruce	-1.4 – 13.4	NPK	Weetman et al. 1987
Average	**6.0**		
Balsam fir	4.4 – 12.9	NPK	Weetman et al. 1987
Average	**8.7**		

* N refers to nitrogen fertilization with urea or ammonium nitrate at various dosages. NPK refers to an array of fertilizers combining some amount of N, P, and/or K.
** Urea was applied six times over a 10-year period to attain optimal nutrition.
*** Eight- and 9-year responses were extrapolated to estimate the 10-year volume increment.

4. Forest Protection

4.1 Forest Fire

Fire plays a prominent role in northern forest ecosystem structure and function and is the primary form of natural disturbance for several of Ontario's major forest types (Heinselman 1981, Wein and MacLean 1983, Kasischke and Stocks 2000). Although fire suppression over the past century has greatly reduced the forest area burned annually, fire remains a major agent of change in the forest. Over the past few decades, the occurrence of forest fire has taken on increased significance due to its influence on global C cycling and human efforts to reduce GHG emissions and associated climatic change (Flannigan et al. 1998, Kurz and Apps 1999, Amiro et al. 2001a). The potential effect of fire in boreal forests is particularly important since this biome covers 1.43 million km^2 and contains 714 billion t C, or roughly 30% of the world's forest area and 37% of the C present in the terrestrial biosphere (Kasischke et al. 1995). Moreover, climate change is expected to result in warmer, drier conditions at higher latitudes where this biome occurs, lengthening the fire season and increasing the area burned above the current level of 5 to 12 million ha yr^{-1} (Wotton and Flannigan 1993, Kasischke 2000).

Since 1930, an average of 1.3 million ha of forest has burned annually in Canada, with up to 7 million ha burning in extreme fire years (Amiro et al. 2001a). More recently, a trend toward higher incidence of fire in the circumpolar boreal forest has been observed. For example, fire frequency was approximately two and a half times higher in the 1980-90s than the 1950-60s (Stocks et al. 1998, Zimov et al. 1999, Amiro et al. 2001a). This increase in fire occurrence and area burned in Canada observed over the past 40 years is due to many factors, including most notably the climatic warming trend that has occurred (Stocks 1991, Kurz and Apps 1999, Kasischke 2000, Saunders 2000, Ward and Mawdsley 2000).

The probability of forest fire is strongly related to climate, stand age, fuel characteristics, forest type, and the intensity of fire detection and suppression efforts (Kurz et al. 1995, Saunders 2000, Amiro et al. 2001a, Ward et al. 2001). Most forest fires in Ontario occur in the boreal regions of the province. Wildfires south of 46° N are negligible (Perera et al. 1998) due primarily to better detection in more heavily populated regions and the predominance of less flammable mixedwood and hardwood forest types in southern Ontario. The number of fires and area burned varied considerably from 1917 to 2000 in Ontario (Fig. 4.1). On average, 1,480 fires burned 140,000 ha each year. During this period, there were 20 severe fire years in which more than 280,000 ha (i.e., twice the provincial average for this period) burned (Fig. 4.1). These severe fire years were centred within two relatively short periods (1917-1933 and 1976-2000), during which large forest areas burned every three years on average. By comparison, from 1933 to 1974, there were only five severe fire years, an average of one every eight to nine years. Below normal forest fire during this period may be linked to relatively cool, wet growing seasons coupled with more effective fire control (Suffling et al. 1982, Kurz et al. 1995). Factors contributing to the recent increase in area burned in Ontario include a warmer and drier climate, increased stand age, and increased mortality by insects and diseases, the latter two causing an accumulation of forest floor and understory fuels (Saunders 2000, Ward and Mawdsley 2000, Amiro et al. 2001a).

Fire and the forest carbon balance

The temporal and spatial changes in forest C pools that occur after stand-replacing fire are species- and site-specific and are strongly influenced by fire severity (Kasischke et al. 1995). Direct, immediate losses of forest C occur from the combustion of living and dead biomass, with the relative loss of C from forest floor, ground vegetation, and the overstory increasing with fire intensity (Ohmann and Grigal 1979, Kasischke et al. 1995, Baird et al. 1999). Most of this direct loss originates from combustion of fine fuels (foliage and small branches) in living vegetation and on the forest floor. In boreal forests, an average of 40 to 60% of the understory vegetation and forest floor is consumed during a fire (for example, Auclair 1985, Kasischke et al. 1995, Levine and Cofer 2000). Larger woody material (living and standing dead overstory stems, fallen boles, and larger branches) burns only partially in a typical fire, and as much as 80% of overstory biomass may remain standing following fire

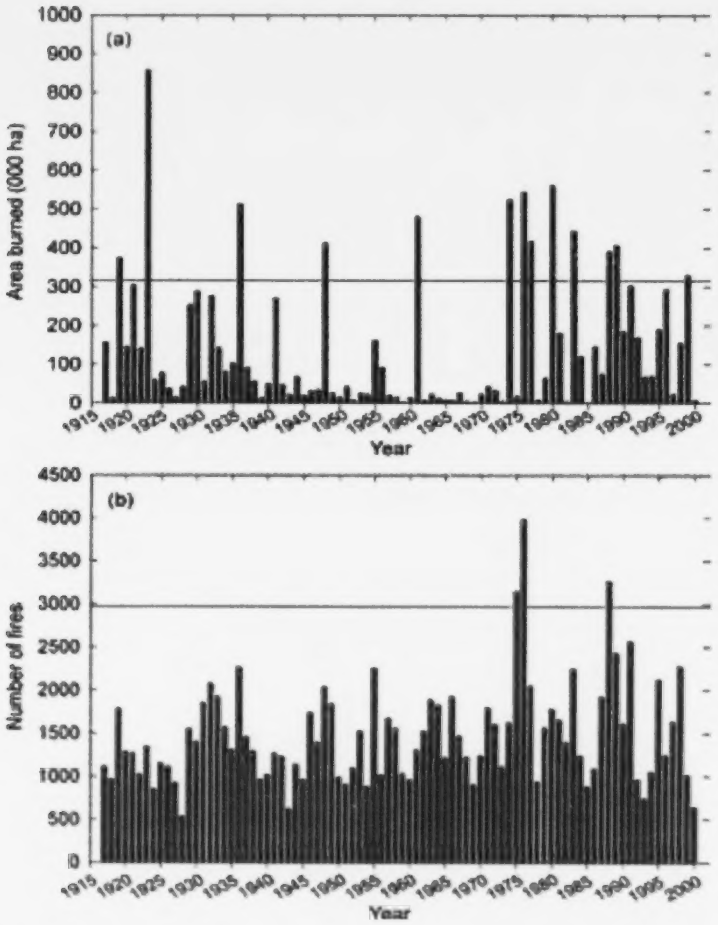

Figure 4.1. Annual area burned (a) and number of fires (b) in Ontario from 1917 to 2000. The horizontal lines represent values twice the mean for this 84-year period.

(Ohmann and Grigal 1979, Auclair 1985). As the primary site of forest C in mature stands, the amount of DOM consumed by fire largely determines the immediate and short-term changes in forest C caused by burning.

The direct and indirect influences of fire on stand C over time are illustrated in Fig. 4.2. Typically, only a fraction of aboveground forest biomass is consumed by wildfire. Direct C losses to combustion range from 7 to 22.5 t ha^{-1}, roughly 9 to 12% of the C in mixedwood and hardwood forests and 20 to 28% of the C in conifer forests (Stocks 1991, Auclair and Carter 1993, Levine and Cofer 2000). In a survey of Canadian wildfires from 1959 to 1999, total annual direct C emissions were estimated at 13 t ha^{-1} (Amiro et al. 2001b). Incomplete oxidation of organic matter by fire also converts a small amount of forest biomass into charcoal. This form of C is very resistant to decomposition with a mean residence time of 10,000 years (Swift 2001). This represents a direct C input (5% of C present) to the DOM and soil C pools (Crutzen and Anderae 1990, Auclair and Carter 1993).

Changes in soil C due to fire depend on the temperature of the burn, the amount and distribution of DOM, and the decomposition rates of residual DOM after the fire. Wildfire may result in total loss of DOM pools where intensity is high (Page-Dumroese et al. 2003). By comparison, fire generally has little direct effect (±10%) on mineral soil C because temperatures in the mineral soil seldom get high enough to cause combustion of organic matter (Ahlgren and Ahlgren 1960, Raison 1979, Johnson 1992a, Johnson and Curtis 2001, Rothstein et al.

2004). Carbon content of upper mineral soil layers may increase slightly if partially burned organic residues and charcoal become incorporated in the soil (Raison 1979, Johnson and Curtis 2001). Where hotter fires occur, they can precipitate a shift of organic matter from upper to deeper mineral soil layers, a redistribution rather than a net loss of soil C (Johnson and Curtis 2001). Contrary to these findings, a recent literature review suggests that soil and DOM C pools of forests are sometimes reduced by wildfire (Hoover 2003).

Erosion following wildfire can result in significant losses of soil C (Elliot 2003). Wildfires in the northwestern U.S. were estimated to reduce soil C by 250 kg ha^{-1}, with about 90% occurring within a year of disturbance. This is about 0.1 to 0.2% of total C present in typical forest soils in this region. Revegetation of burned areas and rapid breakdown of water repellent compounds serves to return erosion rates to pre-disturbance rates. Soil C and sediments are delivered to stream channels or lower slope positions, so this C may either exit the forest ecosystem or be redistributed.

Post-fire C losses to decomposition greatly exceed C uptake by revegetation during the initial phase of stand recovery, with stands generally acting as C sources for at least 20 to 30 years (Fig. 4.2). Reduced canopy shading, decreased thickness of insulating forest floor organic layers, and the darkened ground surface in burned areas elevate surface temperature and accelerate decomposition rates (Kasischke et al. 1995, O'Neill et al. 2002). In northern forests, total post-fire C losses to decomposition are equal to or exceed combustion losses (Auclair and Carter 1993, Dixon and Krankina 1993, Kasischke et al. 1995, O'Neill et al. 2003, Rothstein et al. 2004). Partial thawing of permafrost after fire in more northern boreal forests increases the depth of active soil layers, and further increases decomposition and loss of soil C (O'Neill et al. 2002).

Higher light, moisture, and nutrient levels coupled with warmer forest floor temperatures after fire stimulate the rapid re-establishment and growth of woody and herbaceous vegetation (Ahlgren and Ahlgren 1960, Reich et al. 1990, Hungerford et al. 1991). The change in the forest floor environment and the rate of revegetation depends in part on the amount of forest floor material and vegetation consumed by fire. Stand NPP increases rapidly during this transient period of increased growing space and resource availability, with NPP approaching that of mature stands within 20 to 30 years (Nalder and Wein 1999, Amiro et al. 2000). This initial phase of revegetation is associated with increasing amounts of above- and belowground litter being added to the DOM and mineral soil C pools. Fresh litter and the gradual deposition of burned overstory snags to the forest floor (if not salvaged) also contribute to recovery of DOM C (Auclair 1985, Slaughter et al. 1998, Page-Dumroese et al. 2003). With time, changes in living biomass accumulation and decomposition begin to slow as resources become more limiting, litter production stabilizes, and canopy shading reduces forest floor temperatures (Fig. 4.2).

At the landscape scale, the extent and frequency of fire can strongly influence forest C storage through its effect on forest age-class distribution (Kasischke et al. 1995, Kurz et al. 1995). As fires become more frequent, the amount of forest C present on the landscape also decreases due to the larger number of younger stands with lower C contents (Kasischke et al. 2000). Conversely, a decrease in fire frequency and area burned through fire suppression has reduced the natural fire interval, and effectively increased the forest area occupied by older stand age classes with greater C stocks (Martell 1994, Ward et al. 2001).

Fire has different effects on C storage in hardwood and conifer stands. Fires in hardwood or mixed hardwood-conifer forest are less common, less destructive, and consume less C than fires in conifer forest (Bormann and Likens 1979, Johnston 1995, Amiro et al. 2001a). The rates of ground fire ignition and spread in northwestern Quebec increase from hardwood to mixedwood to pure conifer forests due to the spatial distribution and higher flammability of conifer surface fuels, rather than greater total fuel load (Hély et al. 2000). Fuel characteristics in conifer forests significantly increase the risk of fire (Hély et al. 2000). Hardwood tree species also have faster growth rates, form higher density wood, and accumulate more forest biomass per unit area than conifers (Bormann and Likens 1979, Birdsey 1992, Nalder and Wein 1999, Amiro et al. 2000). Mixedwood and hardwood stands reduce the incidence of fire and increase forest C storage on the landscape (Heinselman 1981, Kasischke et al. 1995, Amiro et al. 2001a, Hirsch et al. 2002).

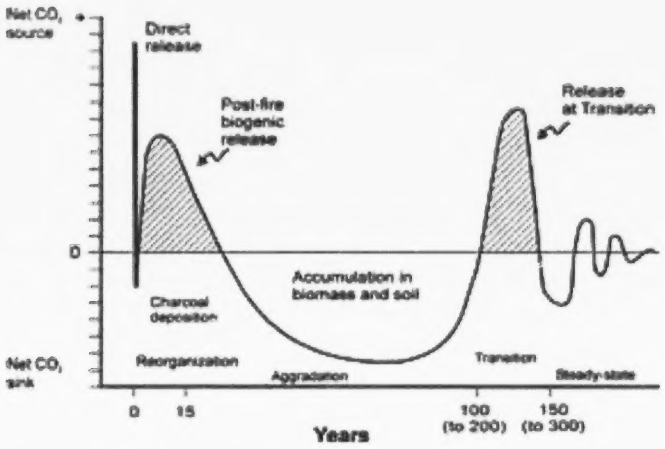

Figure 4.2.
Conceptual diagram illustrating direct and indirect changes in stand C balance following stand-replacing fire (adapted from Auclair and Carter 1993).

Forest fire management in Ontario

The intent of fire suppression in Ontario is to detect, attack, and extinguish wildfire to reduce the forest area burned. Every fire on Crown land receives a response based on the fire's potential impact on people, property, and other values, as well as the estimated cost of the response (Ward and Mawdsley 2000). Three fire management zones are defined in Ontario: extensive, intensive, and measured, each receiving a different level of fire protection for an annual cost of $85 million in 2000 (McAlpine 1998, Ward and Mawdsley 2000, OMNR 2004a)[1] . From 1990 to 2001, extensive fire management was applied to 26.6 million ha in the far north (generally north of 52° N and the Planning Area), where forests are not commercially harvested and settlements are small and dispersed. Forest fire is allowed to play its natural ecological role in this zone, with fire suppression applied only to protect communities and infrastructure (e.g., mine sites, telecommunication facilities). The intensive fire management zone covers areas in northern and central Ontario with major population centres, recreational areas, and commercially used forest. Aggressive detection and suppression of fires within this 40.3 million ha zone is intended to minimize the area burned and the negative impacts on the public and values. Separating the extensive and intensive regions is a narrow band of 11.1 million ha where measured protection is practised. In this area, intensive fire management is applied as soon as a fire is detected. However, if these efforts fail to extinguish a fire, application of further suppression efforts are evaluated relative to the threat to communities and resources.

The comparative influence of intensive, measured, and extensive fire management on forest fire activity in Ontario from 1990 to 2001 is summarized in Table 4.1. With increasing fire management efforts, individual fires are more readily detected and suppressed, reducing both the total area burned and the average size of individual burns. This alters the frequency distribution of the mean number and size of fires annually, with the intensive and measured fire zones being characterized by a higher number of small fires than occur in the extensive management zone (OMNR, Aviation and Forest Fire Management Branch, unpubl. data). As well, the annual burn fraction (ABF) was reduced by more aggressive fire management. From 1990 to 2001, the intensive fire management zone had an ABF of 0.120% compared to ABF values of 0.304% and 0.466% for the measured and extensive zones, respectively (Table 4.1). From these time-averaged ABF values, practising intensive fire management in the current extensive and measured zones could reduce area burned by 92,000 and 20,000 ha yr^{-1}, respectively, or about 112,500 ha yr^{-1} for the entire 77.9 million ha currently managed for fire (i.e., covered by the Forest Fires Prevention Act).

[1] A new forest fire management strategy for Ontario was implemented in spring 2005 (OMNR 2004a). This strategy identifies six fire management zones, based on similar management objectives, land use, fire regime and forest ecology. Within these zones, a continuum of levels of protection can be applied. These levels are referred to as full, modified, and monitored, and are similar in definition to intensive, measured, and extensive protection, respectively, discussed above.

Applying intensive fire management to the measured zone would decrease the average area burned by about 20,400 ha yr^{-1}. Using the values from Table 4.1 and CBM-CFS2 model for Site Class 2 black spruce stands, we predicted the effect of increasing the intensity of fire management on forest C. In this exercise, we assume stands average 100 years old when burned, are regenerated naturally after fire, and unburned stands remain undisturbed over this time frame. From 2008 to 2012, expanding the intensive fire management area to include the 11.1 million ha measured fire zone would result in the storage of an additional 4.1 million t C (Table 4.2). Notwithstanding the increased suppression costs, it is clear that applying intensive fire management to a larger forested area can significantly contribute to forest C storage and fibre yield.

Although expenditures and technological developments have greatly increased the effectiveness of forest fire suppression, each year a small percentage of fires escape initial attack and burn very large areas. From 1990 to 2000 in the intensive protection zone, less than 2% of the more than 13,000 fires responded to in Ontario burned areas larger than 1,000 ha, but these few large fires accounted for 97% of total area burned during this period. Additional investment in fire suppression is not likely to significantly reduce the occurrence of these large destructive fires (McAlpine and Hirsch 1999, Hirsch et al. 2001). Moreover, even if enhanced fire suppression to increase C storage in boreal forests was possible, it can be argued that this approach is impractical, prohibitively expensive, and ecologically undesirable (Kasischke 2000). Alternative, proactive approaches to fire management that reduce the risk of fire are being evaluated (Amiro et al. 2001a, Hirsch et al. 2001). Fuel management is a critical feature of these approaches, with silvicultural practices (e.g., prescribed burning, thinning) and forest management planning (e.g., harvest block layout) being used to decrease the rate of spread and intensity of fire. These approaches could be used at small scales to better protect communities, commercial forests, "enhanced forest management areas", or other special values, but their potential to reduce fire at the provincial scale remains to be demonstrated (Amiro et al. 2001a, Hirsch et al. 2001)

Table 4.1. Mean annual burn fraction (ABF) and number, size, and total area burned annually by wildfire in the extensive, measured, and intensive fire management zones in Ontario (1990 to 2001) (OMNR, Aviation and Forest Fire Management Branch, unpubl. data).

Fire management zone	Area (million ha)	Mean annual fires (number)	Mean annual area burned (ha)	ABF (%)	Mean fire size (ha)
Extensive	26.6	92	123,932	0.466	1,345
Measured	11.1	135	33,816	0.304	251
Intensive	40.3	1,225	48,123	0.120	39

Table 4.2. Total stand C gain from applying intensive fire prevention to the zone of measured fire protection.

Year of fire protection	Stand C in 2012 if not burned	Stand C in 2012 if burned	Loss of stand C (t)	C gain on 43,000 ha (t)
2012	204.9	182.2	22.7	464,000
2011	206.1	172.6	33.5	683,000
2010	207.3	165.5	41.8	852,000
2009	208.5	159.9	48.6	991,000
2008	209.6	155.3	54.3	1,109,000
Total				**4,099,000**

4.2 Tree Diseases and Insect Pests

Insect pests and diseases of trees are important disturbance agents in Ontario's forests, affecting much more area annually than wildfire and harvesting. Insects alone have been responsible for moderate to severe defoliation on about 15 million ha yr^{-1} from 1975 to 2002, with eastern spruce budworm (*Choristoneura fumiferana* Clem.) being the primary defoliator on an average of about 8 million ha yr^{-1} (Fig. 4.3). By comparison, an average of 215,000 ha burned and 212,000 ha were harvested each year during this period. Not only do insects and disease cause mortality and reduce growth, they can delay regeneration, cause stem deformations, predispose trees to windfall, reduce stand density, and increase stand susceptibility to secondary pests (Whitney et al. 1983, Gross et al. 1992).

While insects and tree diseases play important ecological functions, reducing the area affected by these pests would reduce associated mortality and growth losses, and increase timber supply and the amount of C stored in forests (Blais 1983, MacLean 1990). Timber losses to insects and disease range from half (CCFM 1997) to two-thirds (Hall and Moody 1994) of the amount harvested in Ontario each year (Howse 1995). For example, from 1982 to 1987, when spruce budworm populations were at relatively high levels, the total mean annual loss of merchantable volume to forest pests in Ontario was almost 32 million m^3 (Gross et al. 1992). Of this total wood volume loss, about one-third was attributed to insects, 20 to 27% to stem wood decaying diseases, and about 26% to root rot diseases (Gross 1985, Gross et al. 1992). Large annual growth and volume losses to insects and disease, concerns about current and future wood supply, and the potential effects on forest C stocks, will increasingly lead forest managers to consider operational measures to reduce the amount of harvestable timber lost to insects and disease (Whitney et al. 1983, 2001, MacLean 1990, Scarr et al. 2001, Vasiliauskas 2001).

Major forest insect pests and tree diseases in Ontario

Among forest insect pests, spruce budworm is by far the most destructive found in Ontario's forests. Balsam fir exhibits the bulk of the mortality, with lesser damage occurring to white and black spruce. During the outbreak in northwestern Ontario from 1943 to 1955, losses to mortality were about 62 million m^3, equivalent to almost 60% of existing host tree volume (MacLean 1980). Other significant insect pests are defoliators such as jack pine budworm (*Choristoneura pinus pinus* Freeman), forest test caterpillar (*Malacosoma disstria* Hübner), gypsy moth (*Lymantria dispar* L.), hemlock looper (*Lambdina fiscellaria* Guenée), and large aspen tortrix (*Choristoneura conflictana* [Walker]), but their combined impact is generally minor compared to spruce budworm, even in years of low spruce budworm populations (Fig. 4.3, Table 4.3). For example, an outbreak of jack pine budworm from 1982 to 1987 moderately to severely defoliated a total of 7.1 million ha in Ontario (Howse et al. 1995). During this same period, spruce budworm defoliated more than 37 million ha (Howse et al. 1995).

Losses to some insects occur cyclically with the rise and fall in insect populations. Both jack pine budworm and forest tent caterpillar outbreaks erupt every 8 to 10 years, defoliating extensive areas. Peak populations and damage from these species persist for 2 to 4 years and then collapse due to a variety of natural factors (Howse 1995). This boom-and-bust cycling is also characteristic of spruce budworm, which historically has had peak population levels in Ontario about every 35 years, with outbreaks lasting 15 to 20 years (Candau et al. 1998), during which stands are repeatedly defoliated (Fig. 4.3). The relative growth reductions and mortality suffered from defoliation depends on the number of successive years of defoliation and differs among host species (Table 4.4) (MacLean 1990). Under extreme circumstances, the combination of growth reductions and mortality from spruce budworm may decrease stand volume by up to 40 to 80% (McLean 1990).

Stem decay and root rot are the most significant tree diseases affecting Ontario's forests. Of the stem decay diseases, Hypoxylon canker of *Populus* sp., caused by *Entoleuca mammata*, accounted for about half of the total merchantable volume losses in the 1980s (Table 4.5) (Gross et al. 1992), with *Phellinus tremulae* also causing major losses. These diseases also accounted for significant birch and maple volume losses over that time (Table 4.5). Stem decay losses in conifers were most significant in balsam fir, black spruce, and jack pine, with

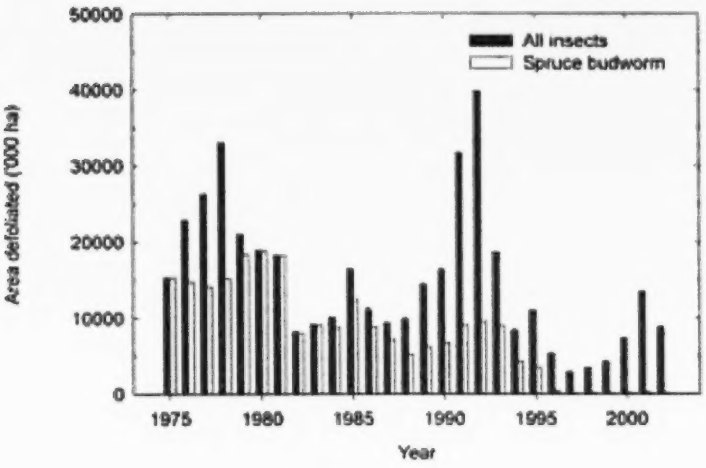

Figure 4.3.
Forest area in Ontario moderately to severely defoliated by all insects and spruce budworm alone from 1975 to 2002. Moderate defoliation is defined as removal of more than 30% of current years foliage. (Data from CCFM 1997.)

Table 4.3. Major forest insect depletions of merchantable volume (millions of m^3) in Ontario from 1994 to 1999 (OMNR 1998, 2000a, 2001a,b,c, 2002b). Total area (millions of hectares) with moderate to severe defoliation (trees with >30% of current years foliage removed) by these insects is also presented.

Agent	1994	1995	1996	1997	1998	1999	Total
Spruce budworm	10.39	11.23	9.93	7.21	5.21	3.92	47.89
Jack pine budworm	0.05	0.07	0.05	0.03	0.00	0.00	0.20
Forest tent caterpillar	0.35	0.00	0.25	0.42	0.56	0.85	2.43
Gypsy moth	<0.01	<0.01	<0.01	0.00	0.00	0.00	<0.03
Total volume loss	10.80	11.29	10.24	7.65	5.77	4.77	50.52
Total area defoliated	4.86	4.01	1.40	1.84	3.13	3.75	18.99

the major diseases responsible being *Phellinus pini, Fomitopsis pinicola, Haematostereum sanguinolentum, Scytinostroma galactinum* and *Inonotus tomentosus* (Gross et al. 1992). Black and white spruce lost the most volume to root rot diseases in Ontario, with lesser amounts of balsam fir and *Pinus* sp. affected (Table 4.5). Pathogens causing root rot are the *Armillaria* complex and *Inonotus tomentosus*, the former increasing after harvest due to the presence of infected stumps and roots (Whitney 1994). The spread of infection of both diseases is facilitated in plantations by heavy machine traffic (Whitney 1994, Whitney and Dumas 1994). *Armillaria* is the most extensive and destructive root disease of forest trees in Ontario, with up to 5% average annual mortality occurring in young spruce plantations (Whitney and Dumas 1994).

In contrast to the cyclical nature of damage by many insect species, losses to disease are usually more constant over time. The exception to the static nature of disease losses results when forest management practices (site preparation, thinning) or natural disturbances (i.e., drought, windstorms, ice storms) reduce tree vigour and/or cause direct physical damage to trees, which makes them more susceptible to disease infection and spread. Quantifying disease losses due to periodic spikes in infection from these stress factors is difficult since there are usually few visual symptoms and, even if the disease results in tree mortality, it usually occurs several years after the causal event. For example, drought, flooding, high temperature, and other abiotic stresses may weaken trees and increase their susceptibility to infection and mortality by *Armillaria* root disease (Wargo and Harrington 1991, Whitney and Dumas 1994).

Table 4.4. Predicted reductions in growth rate caused by spruce budworm defoliation of black spruce, white spruce, and balsam fir stands (from Gross 1985).

Years of moderate to severe defoliation	Reduction in growth rate (%)	
	Black and white spruce	Balsam fir
1	22	0
2	31	18
3	40	35
4	50	57
5	60	81
6	70	91
7	78	

Growth reduction, mortality, and degradation of stem quality due to stain and/or stem decay diseases can result from mechanical wounding suffered during partial harvesting operations (e.g., thinning, single-tree selection) (Nyland and Gabriel 1971, Vasiliauskas 2001). Damage to tree crowns, boles, and roots may occur during felling of trees, movement of harvesting machinery, and skidding of felled stems to landings (Nyland and Gabriel 1971, Vasiliauskas 2001). A literature review of damage and pathology associated with forestry operations in temperate forests indicates that an average of 5 to 20% of residual trees in conifer and mixed stands sustain stem and root damage after a single thinning (Vasiliauskas 2001). However, in North American hardwood stands, damage due to partial harvesting may affect 40 to 60% of remaining trees (Ostrofsky et al. 1986, Nyland 1986, Dwyer et al. 2004). The probability of infection and subsequent growth and wood quality losses due to stain and decay diseases varies among species, and depends on the size and location of the wound and the time since wounding (Lavellée and Lortie 1968, Ostrofsky et al. 1986, Nyland 1986, Anderson and Rice 1993, Vasiliauskas 2001).

Forest management to avoid or protect forests from damaging insects and disease

Avoiding insect and disease damage

Forest managers can reduce damage (i.e., growth reduction and mortality) from insects and disease through activities directed at avoidance. Since many pest problems intensify with stand age (MacLean 1980, Blais 1983, Whitney et al. 1983, Whitney 1995), the selection of a "pathological" rotation for a given site-forest type combination is one way to limit losses. Pathogen damage can also be reduced by proactive silvicultural intervention to create stand conditions that are unsuitable for a given forest pest (Safranyik et al. 1998, Scarr et al. 2001, Vasiliauskas 2001, Whitney et al. 2001, deGroot et al. 2005). Incidence of *Armillaria* root disease in young conifer plantations can be decreased through stump removal and root raking to decrease the amount of infected woody material that serves as inoculum (Hagle and Shaw 1991, Morrison and Mallett 1996). Thinning to increase residual tree vigour can reduce susceptibility to attack by some insect species (e.g., mountain pine beetle (*Dendroctonus ponderosae* Hopk.), jack pine budworm) or increase host tree potential to recover from defoliating insect species (e.g., gypsy moth, spruce budworm) (Mitchell et al. 1983, MacLean and MacKinnon 1997, Safranyik et al. 1998, deGroot et al. 2005). Obviously, stand density management treatments to improve stand vigour and avoid pest damage must minimize mechanical wounding of residual trees (Basham 1993, Safranyik et al. 1999, Vasiliauskas 2001, Whitney et al. 2001).

The probability of stem and root injury and diseases due to logging damage to residual trees by felling and skidding has been strongly linked to proximity to skid trails, tree size, harvest intensity, logging method, and preharvest basal area (Nyland and Gabriel 1971, Ostrofsky et al. 1986, Nyland 1986, Nichols et al. 1994, Fajvan et al. 2002, Dwyer et al. 2004). Depending on site and species, these relationships can be used to minimize associated logging damage and diseases by careful planning of felling and skidding operations, along with training and supervision of workers (Nyland and Gabriel 1971, Nyland 1986, Basham 1993, Safranyik et al. 1998, Vasiliauskas 2001, Whitney et al. 2001). Where injury to residuals is unavoidable, salvage logging of these trees within 10 years can minimize stem decay and degradation of log quality. Where planned stand entries

Table 4.5. Average annual losses in merchantable stem volume from growth reduction and mortality caused by major tree diseases in Ontario from 1982 to 1987 (adapted from Gross et al. 1992)*.

Disease type and tree species	Merchantable volume loss ('000 m^3)	Percent of total mechantable volume
Populus wood decay	4,214	25.8
Birch and maple wood decay	946	5.8
Conifer stem decay	2,478	15.2
Black spruce	722	4.4
White spruce	55	0.3
Balsam fir	710	4.3
Jack pine	803	4.9
Red and white pine	188	1.2
Conifer root rot	8,690	53.2
Black spruce	5,529	33.9
White spruce	393	52.4
Balsam fir	1,973	12.1
Jack pine	667	4.1
Red and white pine	128	0.8
Total	**16,328**	**100**

* Total stand biomass C is estimated from merchantable volume as follows: The average age of black spruce stands in Ontario is about 85 years. Wood volumes (m3) are converted to t of C (1 m3 = 0.25 t C). From CBM-CFS2 simulations, at age 85 the ratio of C in merchantable stem volume to total tree biomass C (above and belowground) is 0.4548. The conversion factor for t of biomass C from m3 of merchantable volume with above average stand age is 0.55 (t of C = 0.55 x merchantable stand volume).

occur within this time frame, selective harvesting can be focused on merchantable stems that sustained injury in the previous harvest.

Susceptibility of forest stands to some pathogens can be altered by modifying species composition. Severity and frequency of spruce budworm outbreaks can be reduced by managing forest tree species composition to increase the hardwood component or by reducing the proportion of balsam fir (Blais 1983, Hardy et al. 1983, Régnière and Lysak 1995, MacLean 1996, Su et al. 1996). Similarly, damage by *Armillaria* root disease can be decreased by favouring more resistant species in planting and early stand density management operations (Hagle and Shaw 1991, Morrison and Mallett 1996). When damage by pests that spread rapidly in single-species plantations with genetically uniform plant material is of concern, promoting age class and species diversity and avoiding large contiguous areas of even-aged monocultures should be considered (Whitney and Dumas 1994, Hubbes and Lin 1996). Currently, strict monocultures are unusual in Ontario given the existing extensive form of forest management. However, future increases in afforestation, intensive forest management, and establishment of bioenergy plantations to mitigate climate change may increase the area of monocultures.

Natural regeneration may moderate losses to insect pests since natural stands tend to be more species diverse and have higher populations of natural insect predators than planted stands (Lethiecq and Regniere 1988, Su et al. 1996). Losses to *Armillaria* are typically higher in plantations than in naturally regenerated stands due to the lack of diversity along with injury and post-planting stress associated with artificial regeneration (Hood et al. 1991). Using healthy, vigorous stock, good planting practices, and matching tree species to site

environment reduces planted seedling stress, therefore avoiding extreme losses to this root disease (Hood et al. 1991). Establishing plantations of eastern white pine significantly increases the probability of mortality by blister rust (*Cronartia ribicola*) since the microclimate of clearcuts favours the spread and infection by this exotic disease (Whitney et al. 2001). Alternatively, regular pruning of lower branches can reduce mortality.

The influence of forest management on susceptibility to insect outbreaks is best exemplified by considering spruce budworm in boreal forests. The frequency and severity of spruce budworm outbreaks has increased in eastern Canada in the 20th century, due to increased fire suppression, clearcutting for pulpwood, and widespread planting of white spruce (Blais 1983, Hardy et al. 1983, Lethiecq and Régnière 1988, Bergeron and Leduc 1998). Ironically, the prolific use of insecticides to control budworm outbreaks in the 1950s contributed to increasing the severity of the 1970s outbreak by inadvertently creating more mature and overmature balsam fir stands (MacLean 1996). These anthropogenic factors have increased the abundance of mature spruce-fir forests, the primary forests supporting spruce budworm outbreaks, on the landscape (MacLean 1980, Blais 1983).

Harvested boreal spruce-fir forests have been converted to hardwood or mixedwood forests due to fire suppression, reliance on unassisted natural regeneration, and inadequate attention to the establishment and growth of conifer plantations (Graham and Knight 1965, Hearnden et al. 1992). The replacement of black and white spruce by mixedwood and hardwood forests reduces their susceptibility to large and long-lived spruce budworm outbreaks (Bergeron et al. 1995, Su et al. 1996). However, any reduction in spruce budworm susceptibility resulting from increased area of boreal hardwood species is likely to be short-lived, since many aspen stands regenerated after conifers are harvested develop a significant balsam fir understory (Graham and Knight 1965). While such stands are resistant to spruce budworm so long as the overstory is present, when the aspen overstory either dies or is harvested and balsam fir advance regeneration is released, the ensuing stands will be susceptible to spruce budworm attack (Graham and Knight 1965). Mortality of aspen stands in northeastern Ontario following an outbreak of forest tent caterpillar has produced just such a result (Candau et al. 2002). It remains to be seen whether the conversion of pure conifer forests to mixedwood and hardwood forests coupled with fire suppression have made Ontario's forests, as a whole, more or less susceptible to future spruce budworm outbreaks.

Protecting forests from insect and disease damage

Forest protection refers to controlling the spread of a damaging insect or disease after it has appeared. Protecting forests using insecticides in Canada has focused in recent years primarily on two species, spruce budworm and hemlock looper. Of the 5.9 million ha treated with insecticides from 1988 to 2003, 66% was treated with the biological agent *Bacillus thuringiensis* (Bt), 31% with fenitrothion, and the remainder with tebufenozide, carbaryl, or other compounds (CCFM 2005). During this period, about 400,000 ha in Ontario were treated for insect control, primarily using Bt. However in the last several years, use of chemical insecticides to control insects in Ontario has been negligible (OMNR 2001a,b,c, 2002b). Aerial application of biological control agents occurred most recently in 1995 and 1996 on about 76,000 ha when Bt was used against a jack pine budworm outbreak (OMNR 1996, 1997).

The best documented example of protection from forest pests in Ontario in the past few decades is the annual aerial spraying of spruce budworm infested areas with insecticides from 1968 to 1987 (Howse et al. 1995). This program focused on protecting commercial and high-value stands (e.g., parks, seed production areas) from defoliation, and in some cases, preventing the spread of the insect into extensive, adjacent areas of susceptible forests (Howse et al. 1995). Nothing has been done to protect merchantable timber from spruce budworm in Ontario since 1996. This inactivity reflects current low spruce budworm populations but may also be due to public sensitivity to pesticide spraying (Carrow 1995, Wagner et al. 1998). Factors such as expense and public acceptability have resulted in a policy that restricts spraying to (1) suppressing insects in outbreak

epicentres, (2) containing outbreaks, and (3) protecting high value or commercially operable forest areas (Carrow 1983). Candidate forests for spraying to control spruce budworm are usually those scheduled for harvest within 10 years that contain spruce and balsam fir exceeding 40% of merchantable stand volume and are essential to the local wood supply. However, past attempts at reducing budworm damage using insecticides have, in some cases, prolonged the outbreaks and reduced the interval between outbreaks (Blais 1983, MacLean 1996). Moreover, given public opposition to the use of even relatively safe insect control agents such as Bt, and the extensive area affected by major spruce budworm outbreaks, spray programs can likely only be used to prevent mortality of particularly valued stands from repeated defoliation. Decisions about the use of chemical spray programs to mitigate losses to provincial wood supply will likely be needed when this issue reemerges during Ontario's next spruce budworm outbreak.

Because of public concerns surrounding biological and chemical control of insect pests in forestry, future management of forest pests will likely emphasize avoidance through silviculture rather than protection (Scarr et al. 2001). As well, forest managers will be increasingly asked to choose "pest battlegrounds" carefully, using attributes such as species mixture, stand age, desired wood products, and types of pest problems to identify stands that would most benefit from some form of pest intervention (McLaughlin and Dumas 1996).

Quantifying losses in merchantable timber and carbon due to insects and disease

The influence of insect damage on timber supply and forest stand C is illustrated by considering spruce budworm activity in Ontario over the past 65 years. Three spruce budworm activity zones are recognized in Ontario (northwest, northcentral and northeast), with the timing of defoliation between zones being slightly out of phase (Table 4.6). Since the last major outbreak began in northeastern Ontario about 1970 and in northcentral and northwestern Ontario about 1980, the next major outbreak in the northeast is imminent, while outbreaks in northcentral and northwestern Ontario are projected for 2015.

Fibre losses from mortality and reduced growth due to a spruce budworm outbreak in northwestern Ontario in the 1940s and 1950s that covered 426 km^2, totalled almost 63 million m^3 in merchantable volume, consisting of about 28 million m^3 balsam fir, 18 million m^3 white spruce, and 16 million m^3 black spruce (Howse 1981). This translates to 34.7 million t of stand biomass C. Large timber losses to spruce budworm also occurred during an outbreak in northeastern Ontario that began in 1966 (Howse 1981). Losses within the 1.8 million ha affected during this outbreak are believed to have exceeded 55 million m^3 of merchantable timber up to 1980 (Howse 1981), of which balsam fir mortality accounted for 29 million m^3 (Howse 1981). By the time this outbreak ended in the late 1980s, loss of 126 million m^3 of merchantable volume (69.3 million t of stand biomass C) was estimated, with 85% due to mortality and 15% associated with growth loss (Gross et al. 1992, Howse 1995). By comparison, merchantable volume losses to spruce budworm during a period of low populations from 1994 to 1999 ranged from 3.92 to 11.23 million m^3 (2.16 to 6.18 million t stand biomass C) when the annual area affected ranged from 0.5 to 4.3 million ha (Tables 4.3 and 4.7). Losses in merchantable volume to budworm during this period totalled 47.9 million m^3, roughly 26.4 million t of stand biomass C. During this same period, reductions in total merchantable volume and stand biomass C due to three other major insects totalled 2.7 million m^3 and 1.5 million t, respectively (Table 4.7).

Significant timber and stand C loss to diseases has also been common in Ontario. For example, merchantable stem volume losses to disease in Ontario from 1982 to 1987 were estimated at about 16.3 million m^3 yr^{-1}, with corresponding losses in biomass C being about 9 million t yr^{-1} (Gross et al.1992) (Table 4.5). Stem wood decay of *Populus* accounted for the loss of 2.3 million t C (26% of the total disease loss) while wood decay of birch and maple accounted for an additional 0.52 million t C (6% of total due to disease). In comparison, annual stem decay losses in conifers totalled 1.4 million t C or 15% of the total loss in stand biomass C (Gross et al. 1992). Annual C losses to root rot diseases in conifers from 1982 to 1987 averaged 4.8 million t, accounting for about 53% of the total C loss to disease (Gross et al. 1992). Annual stem decay losses in conifer species totalled about

1.4 million t C. Losses of biomass C from root rot diseases were most severe in black spruce (3.0 million t C), balsam fir (1.1 million t), and all pines (0.4 million t) (Table 4.5). During this period, total stand biomass C losses due to these stem decay and root rot diseases were 4.2 and 4.8 million t, respectively.

Gains in carbon storage by controlling or avoiding insect and disease damage

Reducing growth losses and mortality from insect pests and disease is a means of increasing wood supply and C storage in standing forests and wood products (MacLean 1990). Calculations of the amount of additional C that can be sequestered into different C pools as a result of insect control (or how much of the existing stored C is maintained by reducing insect pests and disease) must account for the fate of the forest had the losses not occurred. For instance, by reducing mortality caused by spruce budworm, forest C would remain stored in the woody biomass and be slowly transferred to DOM and soil C pools. If, in contrast, the protected stands are lost to fire, C loss from the forest would be high and the anticipated C storage would not occur. Harvesting rather than protecting these stands would also remove this C from the forest and sequester it in various wood products for different periods of time.

A growth loss due to insect defoliation temporarily reduces the rate of biomass C sequestration, which also reduces the long-term transfer of C into DOM and soil C pools. Growth curves used in Ontario are assumed to be based on stands that have undergone defoliation in the past and their yields are taken to reflect growth rates that incorporate insect-caused growth losses. However, the effect of extensive mortality is not accounted for in these growth curves. Stand mortality by insects reallocates C from the living biomass pool to the DOM pool (including standing dead trees). Regeneration then occurs through succession. Salvage harvesting of insect-killed stands seldom occurs in Ontario, but the C dynamics of these stands are similar to normal harvested stands. If insect-killed stands are not salvaged, then the C is reallocated as dead biomass to DOM in much the same way as in burned stands.

Estimating the effect of forest management on C losses to insect pests and diseases requires consideration of C fluxes between biomass, DOM, and soil pools. Reduced damage (mortality and growth loss) from insect pests and diseases would result in a larger live biomass pool, which could mean greater harvested volumes of wood per hectare and, if wood supply needs remained constant, a reduced total forest area harvested. However, a decrease in damage could also reduce coarse woody debris from dying trees and other inputs to the DOM and soil C pools. If the wood available due to avoidance or protection from insects and diseases leads to increased harvest rates, the net effect may be a reduction in total forest C, as more of the C is exported from the site as wood products rather than entering the DOM and soil C pools.

Table 4.6. Area affected by spruce budworm outbreaks in northeastern, northcentral, and northwestern Ontario since 1940. Units of km^2 x years indicate that areas were counted as defoliated from each occurrence during the time shown (from Fleming et al. 2002).

Area/time period	Zone of spruce budworm defoliaton			
	Northeast	Northcentral	Northwest	Total
Zone area (10^3 km^2)	255	66	96	417
Defoliation area (10^3 km^2 x years)				
1941-1963	769	187	426	1,380
1967-1996	1,780	392	610	2,780
Tree mortality area (10^3 km^2 x years)				
1940-1960	85	14	49	148
1973-1996	159	28	54	241

Table 4.7. Biomass C (millions of tonnes) lost due to mortality and growth reductions caused by four insect species in Ontario from 1994 to 1999 (merchantable stand volumes from Table 4.3 converted to total stand C using a factor of 0.55 C/volume).

	Year						
Defoliator species	1994	1995	1996	1997	1998	1999	Total C loss
Spruce budworm	5.71	6.18	5.46	3.97	2.87	2.16	26.35
Jack pine budworm	0.03	0.04	0.03	0.02	0.00	0.00	0.12
Forest tent caterpillar	0.19	0.00	0.14	0.23	0.31	0.47	1.34
Gypsy moth	0.01	0.01	0.01	0.00	0.00	0.00	0.03
Total	**5.94**	**6.22**	**5.63**	**4.21**	**3.18**	**2.63**	**27.81**

Spraying to prevent stand mortality during the next spruce budworm outbreak in Ontario could affect C storage in forests and wood products. Most of the mortality following repeated years of defoliation occurs to balsam fir (Gross 1985) rather than black and white spruce. Insecticide spraying would most likely occur in stands of high value or commercial operability, and either in insect outbreak epicentres or needed to contain an outbreak, will be sprayed. So, stands with large components of balsam fir, which is of low economic value in Ontario, or spruce stands that are not commercially operable would not be sprayed. Spraying would, in most cases, be done to preserve timber for harvest and, since harvest removes more C from the site than is lost from a non-salvaged stand killed by spruce budworm, spraying would indirectly reduce forest C storage.

Moderate to heavy spruce budworm defoliation reduces growth by 22% after one year to as much as 78% after seven years of repeated defoliation in black spruce and white spruce, and from 0 to 100% over the same periods in balsam fir (Table 4.4) (Gross 1985). Maximum growth rate for Site Class 2 black spruce stands occurs between 40 to 60 years of age when, according to our simulation, approximately 1 t C ha^{-1} yr^{-1} accumulates in total stand biomass. For every successive year of heavy spruce budworm infestation, annual growth declines from 1.0 t C ha^{-1} to 0.78, 0.69, 0.60, 0.50 and 0.40 t C ha^{-1} (Gross 1985). If spraying a black spruce stand once prevented moderate to severe defoliation by spruce budworm for five years, then the additional C sequestered because of spraying would be 0.22, 0.31, 0.40, 0.50 and 0.60 t ha^{-1}. Thus, an average Site Class 2, 40- to 60-year-old black spruce stand protected during five years of a severe spruce budworm outbreak would sequester an additional 2.03 t C ha^{-1} in total above- and belowground biomass compared to a repeatedly, severely defoliated stand.

If during a hypothetical outbreak 60,000 ha of black spruce stands (an area this large was treated in Ontario in the 1990s) were sprayed with insecticide each year from 2008 to 2012, and one treatment prevented defoliation for the ensuing five years, then treatment in 2008 would allow an additional 121,800 t C to be stored by 2012. Spraying a further 60,000 ha in 2009 would allow an additional 88,800 t C to be sequestered by 2012, with 55,800, 31,800 and 13,200 t C sequestered from treatment of 60,000 ha in 2010, 2011, and 2012, respectively. Thus, protecting 60,000 ha yr^{-1} from severe defoliation by spruce budworm would cumulatively sequester 311,400 t C by the end of 2012.

At the provincial level, lower rates of mortality and growth losses due to damage by insects and disease would allow the same amount of wood currently harvested to be obtained from a smaller area. This would increase total C stored in Ontario's forests. Alternatively, an expanded wood supply would be provided by the increase in forest productivity due to forest protection. Reducing the severity of spruce budworm outbreaks would also reduce the risk of forest fire that often follows budworm infestations (Fleming et al. 2002), further increasing wood supply and C storage. If actions were taken to reduce insect- and disease-related growth losses by 10%, then the gain in C stored in Ontario's forests during a period such as the 1990s (when pest problems were relatively low) would be about 1.4 million t C and 2.5 million m^3 of merchantable volume annually (Tables 4.3, 4.5, 4.7).

5. Harvesting and Related Practices

5.1 Roads, Skid Trails, and Landings

Forest management requires access to the forest and space to work within it. This is achieved by establishing a network of temporary and permanent roads for forest access, landings to stockpile and process logs, and skid trails to move logs from within stands to landings. Creation of these areas to support the use of heavy machinery in harvesting and silviculture affects forest productivity and C sequestration both immediately and in the longer-term (Page-Dumroese et al. 2000, Lal 2003). The primary immediate effect on productivity of permanent road construction results from loss of forest area. For secondary and tertiary haul roads with a limited period of use, and for landings and skid trails that are allowed or encouraged to revegetate, soil disturbance from machine traffic is commonly associated with reductions in growth and site productivity (Lal 2003). Soil compaction due to heavy equipment traffic may persist for decades, depending on the natural processes that affect soil recovery rates (Greacen and Sands 1980, Corns 1988, Kozlowski 1999, Lal 2003). Recognition of these impacts has led to increased efforts by some jurisdictions to rehabilitate these areas to diminish any adverse effects and promote recovery to preharvest condition (McNabb 1994).

Mechanization of forest management over the past 50 years has increased the proportional area and severity of soil disturbance, i.e., compaction, rutting, and displacement. However, increasing intensity of forest management does not necessarily result in a commensurately large increase in the network of roads, landings, and skid trails. Usually, the access network created at the time of harvesting to extract timber is also used for subsequent silvicultural work or fire protection. For example, a forest that is harvested, site prepared, and planted is accessed three times using the same road systems. Other management activities, such as the aerial application of herbicides, do not require access roads. Thinning, by comparison, may introduce a second network of roads and skid trails later in the rotation. Reuse of original skid trials for these subsequent interventions will help minimize the area of disturbed and compacted soil per rotation.

Road and skid trail location and harvest block layout should strike an appropriate balance between site disturbance and the need for efficiency (Archibald et al. 1997, Henderson 2001). Achieving this balance requires good communication among the forester, road builder, and logger to ensure that the minimum length of travel corridors needed for safe and efficient harvesting is constructed (Dwyer et al. 2004). Planning road, skid trail, and landing locations to minimize total disturbance requires considerable knowledge of local site conditions, the ecological impacts of soil disturbance, and the need for efficiency of harvesting operations. Using this approach, skid trails and landings can be kept to 10% or less of the harvested area (Nyland and Gabriel 1971, Dwyer et al. 2004).

Maintenance of forest soil productivity by minimizing the area disturbed (e.g., soil displacement, compaction) by harvesting activities in North America is currently addressed through guidelines, standards, or "best management practices" adapted to soil types sensitive to these disturbances (Heninger et al. 1997, Page-Dumroese et al. 2000, Swift 2001). In British Columbia, the Forest Practices Code stipulates that no more than 7% of the harvested area be occupied by roads and landings (Henderson 2001). The code also limits total soil disturbance (i.e., roads, landings, and skid trails) to no more than 10% of the area harvested on sites at high risk of soil compaction (Henderson 2001). There, skid trails are counted as 'disturbed soil' if they are rutted, have had repeated traffic, or have been bladed (Henderson 2001). In British Columbia, roads, landings, and skid trails not needed for long-term management of an area are required to be rehabilitated (Bulmer et al. 1998). In Ontario, no such regulations exist, and statistics on site disturbance by harvesting are not collected. Instead, OMNR has guidelines for protection of the physical environment that are considered "best management practices" and applied based on site factors (Archibald et al. 1997). These guidelines list management practices to be considered when developing and implementing a forest access strategy to minimize site damage. However, Ontario's guidelines provide no precise standards for performance, meaning that identifying 'acceptable' levels of site disturbance relies on forest company and government representatives agreeing on qualitative rather than quantitative criteria.

Soil disturbance by harvesting activities

For purposes of this report, soil disturbance by harvesting activities consists primarily of compaction, rutting, and displacement that may result from felling, extracting, piling, and transporting logs from the cutover site to the mill (for further discussion of the effects of soil displacement, see Section 2.1). Compaction is the increase in soil density in response to repeated, short-duration application of external forces by movement of harvesting equipment over the soil. The amount of compaction varies with the type of machinery used, but generally increases with the volume harvested per unit area and the number of passes made by skidding equipment (Lull 1958, Hatchell et al. 1970, Helms and Hipkin 1986, McNabb et al. 2001). The susceptibility of soils to compaction depends on many factors, including the applied force; the texture, organic matter and moisture content, structure, and bulk density of the soil; and the thickness and composition of the surface organic layers (Lull 1958, Graecen and Sands 1980, Soane 1990, Arnup 1998, Kozlowski 1999). The likelihood of compaction from machine traffic is comparatively higher when soils are wet, finer textured, and low in organic matter (Hatchell et al. 1970, Greacen and Sands 1980, Edwards and Ross-Todd 1983, Kozlowski 1999). Soil compaction reduces the volume of macropores, increases the volume of micropores, and increases bulk density. Changes in soil porosity and bulk density reduce water permeability, decrease gaseous flux, and increase thermal conductivity of the soil (Greacen and Sands 1980, Ruark et al. 1982, Arnup 1998, Kozlowski 1999, Shestak and Busse 2005). The resulting increase in resistance to root penetration can adversely affect regenerating trees by reducing root growth, root system size, water and nutrient uptake, and growth potential (Greacen and Sands 1980, Kozlowski 1999).

Rutting refers to destruction of soil structure caused by deformation of the soil surface. This occurs when soil water content is near or at saturation, and the external force exceeds the shear strength of the soil. Repeated rutting and destruction of soil structure results in "puddling", visually apparent as "soil soup" in areas of heavy machine traffic (Arnup 1998). Rutting is more prevalent on fine-textured soils due to their plasticity and high soil moisture content, and generally occurs during the frost-free period when soils are near saturation. Wetter organic soils are also susceptible to rutting. Rutting affects the physical and biological properties of soils similarly to compaction but adds soil displacement and mixing. Harvesting during the winter when soils are frozen can significantly diminish the incidence of rutting and compaction.

Effects of skid trails on forest growth

Where roads and landings reduce the forested area, their effects are obvious. In comparison, since skid trails are usually revegetated, their effects on forest productivity and C storage are less apparent. Although skidding can result in visible site damage (e.g., rutting), skid trails affect stand productivity and forest C storage primarily through soil compaction. Given the complex interactions between soil properties and tree species, compaction can either increase or reduce individual tree and stand growth (Miller et al. 1996, Kozlowski 1999, Siegel-Issem et al. 2005). Most of the influence of compaction on tree growth along skid trails is caused by changes in site moisture conditions (Greacen and Sands 1980). Compaction can improve tree growth when reduced soil porosity increases moisture retention and maintains adequate aeration in well drained, coarse-textured soils (Conlin and van den Driessche 1996, Zou et al. 2000, Brais 2001, Gomez et al. 2002, Ponder 2004). By comparison, soil compaction negatively affects growth when anaerobic conditions develop. Because even a few roots in a zone of compacted soil can disproportionately affect tree growth, differentiating when skid trail coverage causes positive or negative effects on stand growth is difficult. Undoubtedly, the positive effects of skid trail compaction on moisture availability on coarse-textured soils balances out some of the negative effects of compaction on fine-textured soils.

Under some conditions, compaction of finer-textured forest soils markedly reduces site productivity (Froehlich and McNabb 1984). For example, in a 32-year-old Douglas-fir plantation in the Pacific Northwest, trees growing on compacted soils had 30% less height growth and 55% less volume growth than trees growing 3 m or more from skid trails (Wert and Thomas 1981). In a 16-year-old ponderosa pine (*Pinus ponderosa* Dougl. ex P. & C.

Laws.) plantation established on a clearcut site prepared using a tractor, greater bulk density on and near skid trails increased mortality of planted trees and reduced growth of survivors by 29% (Helms and Hipkin 1986).

Where soil compaction is of concern due to site factors, prevention rather than rehabilitation is most effective. Adverse effects of soil disturbance can be minimized by rating site and soil types for susceptibility to compaction, developing special measures for sensitive sites, considering seasonal variation in site sensitivity, adapting machinery to site sensitivity, and pre-determining travel routes (Hatchell et al. 1970, Martin 1988, Reisinger et al. 1988, Archibald et al. 1997, Heninger et al. 1997, Arnup 1998). On sensitive sites, restricting machine movement to a limited number of skid trails may reduce the portion of a stand affected by compaction as well as damage to advance regeneration and residual trees in partial cutting systems (Hatchell et al. 1970). However, on sites less sensitive to compaction, less damage occurs when skidding is distributed widely (Hatchell et al. 1970, Archibald et al. 1997). Compaction of lightly used skid trails is similar to heavily used ones because most compaction occurs during the first few passes of the machine (Hatchell et al. 1970, Froehlich and McNabb 1984, Kozlowski 1999, McNabb et al. 2001). Any associated negative effects on productivity can be minimized by confining all initial and subsequent entries to the same few trails (Murphy and Firth 2004).

Winter harvesting on frozen soil minimizes compaction, particularly on sensitive sites with wet or organic soils. Where winter harvesting is not possible, harvesting using preplanned skid trails and haul roads is the best option to reduce productivity losses from soil compaction (Froehlich and McNabb 1984, Egan 1999). Preplanned skid trails can economically be held to about 10% of the harvest area and can serve as access routes for all succeeding silvicultural interventions (Froehlich and McNabb 1984). By comparison, skid trails that minimize timber extraction costs or that are freely selected can range from 18 to 40% of the area (Hatchell et al. 1970, Reisinger et al. 1988, Morris and Miller 1994, Buckley et al. 2003). Running skid trails within a cut block on areas of higher load bearing capacity and strengthening load-bearing by covering the ground with matting made of logging slash reduces compaction (Archibald et al. 1997). Using high flotation tires on skidders also reduces the risk of compaction, and in Ontario these should be used when summer logging occurs on sites with organic soils (Archibald et al. 1997). However, even skidders with high flotation tires can compact high traffic areas on susceptible soils (Henderson 2001).

Unaided, the natural recovery of skid trails from compaction may take decades, depending on the severity of compaction and the complex interaction between climate, soil properties, and management practices (Hatchell et al. 1970, Greacen and Sands 1980). The major factors affecting recovery rate are usually the soil type and soil moisture content at the time of skidding. The combination of freezing and thawing, moisture/temperature changes, activity of soil biota, and plant root penetration and decay can contribute to recovery of macroporosity (Reisinger et al. 1988, McNabb 1994). While soil freezing and thawing can reduce bulk density in the upper 5 cm of the soil within a few years, compaction at lower depths can remain largely unchanged for up to 30 to 40 years (Greacen and Sands 1980, Wert and Thomas 1981).

Effects of landings on forest growth

Harvested trees are collected at landings, which are usually located at main road access points and constructed by harvesting standing timber and removing stumps, organic horizons, and surface soil layers (Carr 1987, McNabb 1994). In cut-to-length or tree-length harvesting, trees are delimbed on site, skidded to landings, and stacked until loaded onto trucks for transport to mills. With full-tree harvesting, the trees are delimbed at the landing, where branches and unmerchantable tops of trees then accumulate. Machinery moving over these piles of logging debris can form a compacted woody mat that is slow to revegetate. Regardless of harvesting method, disturbance by heavy equipment in the construction and use of landings significantly compacts soil and reduces site productivity.

Helms and Hipkin (1986) examined the relationship between soil bulk density and growth of ponderosa pine planted on landings and observed that landings had higher bulk density (>1.2 g cm^{-3}) than skid trails and also higher seedling mortality (60%) and poorer growth of survivors (22% less volume). Volume per hectare was

reduced by 69% compared to areas of the stand with the lowest bulk density (i.e., least compacted). Similarly, height growth reductions of 30 to 60% were exhibited by lodgepole pine seedlings 6 to 11 years after being planted on winter and summer landings in British Columbia (Carr 1987). This decrease in seedling growth was attributed both to increased soil compaction as well as reduced nutrient availability on landings associated with the removal of organic matter and topsoil during landing construction (Carr 1987).

According to Hall (1996), forest landings in Canada usually occupy 5 to 8% of the area accessed for harvesting. This is substantially higher than the average estimate of about 1% obtained in a survey of forest managers in northwestern Ontario done by the authors for this report. In Ontario, guidelines to prevent the loss of productive forest do not stipulate a maximum area for landings. Instead, they aim to minimize the area in landings by recommending the following practices (Archibald et al. 1997): (1) locating landings on non-productive areas (e.g., exposed bedrock), (2) using cut-to-length or tree-length logging to reduce roadside slash piles, (3) placing logs in higher as opposed to wider piles, and (4) transporting wood to mills as quickly as possible to minimize the landing residence period. Although it is recognized that "the amount of land lost to roads, landings and slash piles will be significant…Common sense should dictate when techniques employed are resulting in more productive land being lost than is necessary…Minimizing roads and landings may increase logging costs. Therefore, operators need to be informed as to why this is beneficial in the long term" (Archibald et al. 1997).

Rehabilitation of skid trails, landings, and temporary roads

Rehabilitative treatments intended to reverse effects of compaction and alleviate negative effects on tree growth (Kozlowski 1999) should be site-specific since generic application of treatments may further degrade the affected areas (Bulmer 1998, Maynard and Senyk, 2004). Generally, rehabilitation reduces soil compaction through tillage. Mechanical site preparation can loosen the soil along skid trials. In the case of landings, tillage may be followed by the retrieval and redistribution of displaced topsoil, organic material, and woody debris (Hall 1996, Heninger et al. 1997). Fertilization in combination with planting or seeding with native herbaceous species have also been used to rehabilitate landings (Reisinger et al. 1992, McNabb 1994, Bulmer 1998). However, on coarser-textured soils where compaction has improved moisture retention, tree growth, and site productivity, rehabilitative efforts may diminish or reverse these benefits (Foil and Ralston 1967).

Forest companies may consider restoring temporary forest roads and landings to productive forest after logging. Guidelines for rehabilitating landings in Ontario do not exist (OMNR 2004b). Although landings are often replanted, they are seldom treated to reduce compaction or restore nutrient-holding topsoil. As a result, old landings are likely nutrient deficient and support slower tree growth. Without mechanical cultivation and other ameliorative treatments, attempts to reforest compacted landings will never be fully successful and, even if survival rates are acceptable, tree growth will be greatly reduced. Due to the high cost of rehabilitating landings, it may be more cost effective to minimize their construction.

Effects of roads on forested area and forest growth

The construction and presence of roads can have wide-ranging, largely negative ecological effects on terrestrial and aquatic ecosystems (Forman and Alexander 1998, Trombulak and Frissell 2000, Gucinski et al. 2001). Major ecological effects of forest roads are: (1) loss of forest and wetland habitat, (2) altered physical and chemical environment, (3) increased soil erosion and sedimentation of aquatic areas, (4) fragmented habitat and altered patterns of flora and fauna, (5) increased human access and associated disturbance, and (6) introduction of alien/invasive species (Forman and Alexander 1998, Haskell 2000, Trombulak and Frissell 2000, Gucinski et al. 2001, Elliot 2003). Some of these effects may extend several hundred metres from the road corridor. Proper road design and construction can minimize many of these adverse effects (Gucinski et al. 2001).

Forest roads are generally constructed by removing vegetation, including stumps and organic and surface mineral soil layers, to expose subsurface soils. For primary, permanent roads, gravel may be added to the surface to improve transportability and increase longevity. Forested land lost or affected by road construction includes: (1)

the travelled surface, (2) the right-of-way cleared on either side of the road surface, (3) "pushouts" where slash, stumps, and organic matter are deposited in piles off the right-of-way, (4) pits where road building materials such as gravel and sand are obtained, (5) loading areas, (6) turnarounds, and (7) pioneer roads that serve as entry points into forests for road building machinery (OMNR 1995). The right-of-way created on either side of a road to improve visibility may also be graded to slow the re-growth of woody plants. In addition, when a road is constructed, the nutrient-rich, surface organic matter is removed and piled to either side of the right-of-way. Right-of-ways will gradually be revegetated, although as long as the road is in use, trees will be removed from it.

In addition to direct losses of forest to roads, indirect losses in forest productivity may result when roads redirect water (Henderson 1995). Roads that run across slopes (even gentle slopes) interrupt the flow of water and divert it into ditches and streams. This diversion of water from stands can be beneficial after harvesting if it reduces the occurrence of raised water tables and flooding that can occur when the canopy of transpiring trees is removed. However, Henderson (1995) speculates that when moisture is limited, the redirection of water into streams could increase the incidence of moisture deficiencies and thereby reduce forest productivity on the lower side of roads. In some areas of Ontario, topography is sufficient that roads can alter surface and subsurface water flow enough to affect site moisture. However, most of the managed forest in Ontario has very low topographical relief and the hydrological effects of roads should be negligible. In the absence of data, quantifying the effects of water diversion on forest growth is difficult, but intuitively it is expected to reduce growth potential. Roads may also affect forest growth and C storage indirectly if access increases human-caused fire ignitions and more forest area burns. Alternatively, improved access should assist forest fire management, potentially decreasing the area burned.

Estimating the area occupied by skid trails, roads, and landings

In British Columbia, the Forest Practices Code limits road and right-of-way to 7% of the area harvested (Henderson 2001). For tertiary roads averaging 12 m in width, 7% road occupancy is reached when 5.8 km of road are constructed per km^2 of forest harvested. Primary and secondary roads have right-of-ways much larger than 12 m and the 7% threshold occurs at a lower total road length. By comparison, Ontario does not regulate the length of road permitted per hectare of forest accessed, although it would be expected that road density (e.g., km km^{-2}) would be minimized due to the high cost of road construction and maintenance. Forest road right-of-ways are supposed to be at least 20 m wide, except within 100 m of a water crossing where they are to be as narrow as possible (OMNR 1995). Guidelines to prevent the unnecessary loss of productive land in Ontario advocate but do not require the following practices: (1) locating roads to maximize cost-effective skidding distance, (2) avoiding the use of loop turnarounds for haul trucks, (3) planning harvesting to use winter roads when possible, and (4) minimizing right-of-way widths (Archibald et al. 1997).

A road plan forms part of every forest management plan in Ontario. Three classes of forest access roads are recognized in the OMNR's Forest Management Planning Manual (OMNR 2004b). Primary roads provide access to the FMU as a whole. They are permanent roads; i.e., life expectancy more than 15 years. Secondary roads branch from primary roads to access operating areas within a management unit, are not considered permanent, and have a 5- to 15-year life span. Tertiary roads provide access for harvest and renewal operations for up to 5 years. They may be unsurfaced or thinly surfaced and are often replanted or left to regenerate naturally when abandoned. Forest management plans in Ontario must summarize new and existing primary and secondary road construction, their lengths, and the timing of planned abandonment. The Forest Management Planning Manual for Ontario does not mention documenting tertiary road construction (OMNR 2004b); however, in northwestern Ontario, estimates of the area covered by tertiary roads and landings are typically included in forest management plans.

For this report we surveyed OMNR staff in some northwestern Ontario districts in 2001. They indicated that the area in tertiary roads and landings varies from about 2 to 4% of forested land. For example, in the Red Lake Forest Management Unit, aerial photography revealed that 4 to 5% of the forest is displaced by secondary and tertiary roads and landings on upland sites logged during the frost-free season. This is reduced to 2 to 3%

in areas harvested in winter when the soil is frozen, especially on lowland sites (D. New, 2001, OMNR, pers. comm.). Assuming that primary roads account for an additional 2% of forested area, the total area in landings and forest access roads of all types would average about 4% on winter harvest areas and 7% on upland sites harvested in summer.

Most of the road network in the managed forest in Ontario is permanent. Although tertiary roads have a short life span and are abandoned, they are not truly rehabilitated and do not necessarily return to equally productive forest, at least in the short-term. Even though abandoned tertiary roads may support vegetation and be replanted, they are not as productive as other forest land, and increasingly are being used to access second-growth forest. As well, new tertiary roads are continually being built to access previously unharvested tracts of forest resulting in an ongoing loss of significant areas of productive forest through construction of primary, secondary, and tertiary roads.

Forest carbon changes due to skid trails, roads, and landings

The lack of adequate information about the amount and location of skid trails, roads, and landings in Ontario makes it difficult to predict their potential short- and long-term effects on forest C. A complete accounting of past and new deforestation by roads and landings and the areas covered by skid trails is needed to quantify the loss of forest area. However, based on our limited knowledge of the extent of these features, we can still make some rough, qualified estimates on the potential losses of forest area and C associated with these components of forest access. We also assume that losses of soil C due to erosion from these areas are insignificant (< 0.5 kg ha^{-1}) where proper care is taken in their planning and construction (Elliot 2003).

Skid trails

The forest area affected by skid trails in Ontario is unknown, but is likely 10 to 30% of the area harvested. A recent literature review of the effects of forestry-related soil disturbance suggests that soil compaction will decrease forest growth and yield by 5 to 50% (Arnup 1998). It can be assumed that skid trails will reduce forest growth and C storage on sites with fine-textured soils, but compaction may increase growth and C storage on sites with coarse-textured soils. Where significant amounts of coarse fragments occur in the soil, compaction may have no net effect on forest growth or stand C. In addition, because of the influence of soil texture on compaction, the effect of skidding varies with soil moisture content and whether the soil is frozen at the time of skidding. Therefore, we consider that there is no net effect of soil compaction on forest growth and C storage along skid trails.

Roads and landings

The total area of roads and landings in managed forests in Ontario is unknown. Relying on information gathered from staff from several forest management units in northwestern Ontario, we assume that harvesting upland sites in the frost-free season causes a loss of about 7% of forest to roads and landings, while roads and landings created during winter harvest cover about 2% of the forest. Therefore, on average, we assume harvesting converts 4% of the forest area to roads and landings. Some additional forest would be lost to road construction-related features such as gravel pits, turnarounds, pushouts, and pioneer roads, but these areas are not accounted for here. Given approximately 200,000 ha average annual forest area harvested since 1990, and assuming existing primary roads occupy 2% (4,000 ha) of this and that construction of new roads and landings deforested another 2%, about 4,000 ha of forest have been lost due to deforestation to build secondary and tertiary roads and landings annually since 1990, or a total of 60,000 ha from 1990 to 2004.

Effects of roads and landings on stand carbon dynamics

The effect of road and landing construction on stand C dynamics was simulated using CBM-CFS2 (Fig. 5.1). In this scenario, changes in total stand C are estimated for a black spruce plantation established after

clearcutting as well as the area deforested for roads and landings. The difference in C content is also presented. Total stand C at the time of harvest was about 205 t ha^{-1}. An exponential decline in C due to road and landing construction results primarily from litter decomposition and the absence of significant revegetation, approaching a fairly constant value of 80 t C ha^{-1} at 60 years (this scenario assumes no change in humified soil C when in fact deforestation causes a slow, long-term loss from this pool). The difference in total stand C between the plantation and area occupied by roads and landings increases linearly over time and totals 116 t ha^{-1} at 80 years, a typical rotation. As a result, the plantation contains about 2.4 times more C at rotation than the area converted to roads and landings.

This simulation can be used to estimate future forest C losses as a result of roads constructed in 1990. Again, assuming approximately 4,000 ha are converted to roads and landings in 1990, the exponential decline in C associated with road construction results in a total loss of C of almost 102,000 t by 2012. Extending this simulation to a scenario of continued deforestation due to road and landing construction at 4,000 ha yr^{-1} from 1990 to 2012, an estimated loss of about 960,000 t C occurs over this 22-year period. Further, if the area of new roads and landings could be reduced by 10% beginning in 2005, 81,400 t C would still be lost, but an additional 9,000 t C would be stored in forests by 2012 (Fig. 5.1).

Extending forest management beyond 51° N as is presently proposed will require the construction of new access roads, resulting in additional deforestation. About 4.5 million ha of productive forest are north of the Planning Area. Suppose that 10,000 ha of this area were clearcut each year beginning in 2005, 4% (or 400 ha) of which was converted to forest access roads and landings. Using data from the simulation presented in Fig. 5.1, 5.4 t C ha^{-1} will be lost after 7 years, for a total release in 2012 of about 2,200 t C from the harvested area. If each year from 2005 to 2011, 10,000 ha are harvested and 400 ha are converted to roads or landings, then the total amount of C lost by 2012 would be about 9,000 t.

Over the longer term, how much C could be lost to roads and landings because of harvesting north of the Planning Area? By 2070, the area deforested by roads and landings in 2005 would contain 84.1 t C ha^{-1}, or 93.4 t C ha^{-1} less than the clearcut and planted area (177.6 t C ha^{-1}), or about 37,372 t C less over the entire harvested area. If 10,000 ha of new forest were harvested annually from 2005 to 2055 (a total harvest of 500,000 ha over 50 years), new road building and landings will deforest 400 ha yr^{-1} and 20,000 ha in total. By 2070, this would reduce C storage by about 1.1 million t. If the annual rate of deforestation due to roads and landings could be reduced by 10% (from 400 to 360 ha), this would reduce total C loss in 2070 to about 990,000 t, a gain in C stored of about 110,000 t.

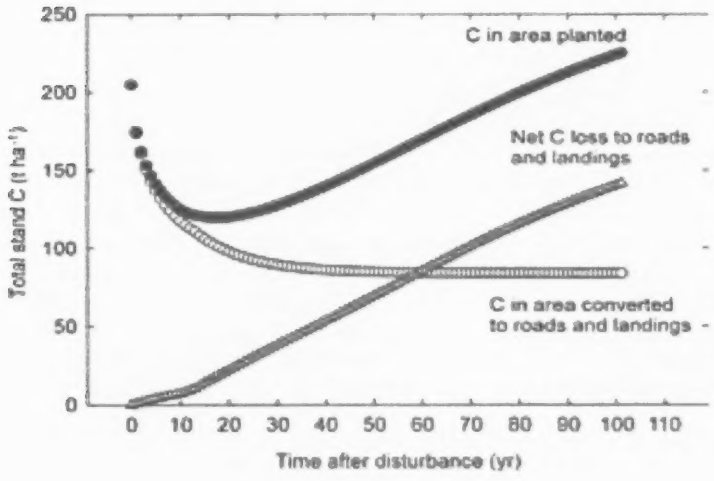

Figure 5.1.
Simulated change in total stand C in clearcut and planted land and areas deforested by road and landing construction for a Site Class 2, black spruce stand. The difference between these two areas is the cumulative loss of C due to road and landing construction.

5.2 Forest Disturbance by Harvesting

Harvesting forest biomass to produce wood products is the primary form of human disturbance in Ontario's forests. From 1990 to 2003, between 180 and 224 thousand ha of forest were harvested each year in Ontario, an average of about 206,000 ha yr^{-1} (CCFM 2005). From 1996 to 2000, an average annual wood volume of 22.4 million m^3 was harvested. Currently, 90% of the area and 95% of the volume harvested annually come from the Boreal Forest Region; roughly 75% of that was conifer (OMNR 2002b). Annual allowable cut, which determines harvest levels, remains relatively constant. By comparison, the cumulative forest area disturbed by natural factors (e.g., fire, insects) varies through time, but is typically much larger than that harvested (Sections 4.1 and 4.2). Wood volume lost annually to all forms of natural disturbance, both from mortality and growth reduction, can be as high as 2 to 3 times that harvested each year (Table 5.1).

The clearcut, shelterwood, seed tree, and selection silvicultural systems are used to manage the harvest of Ontario's forests. These systems differ primarily in harvest intensity, i.e., the proportion of overstory trees that are harvested. Clearcutting is by far the most popular silvicultural system in Ontario and is used to harvest about 88% of the total area and more than 90% of the total wood volume each year. It is used primarily for harvest of jack pine, spruce, white birch, and aspen, forest species adapted to stand-replacing natural disturbance by wildfire and insects. Clearcutting in Ontario refers to the removal of most tree stems, leaving a minimum of 25 residual individual trees ha^{-1} as well as small patches of uncut forest over a small percentage of the area (OMNR 2001d). In 2002, clearcuts in the Boreal Forest Region averaged from 39 to 1,986 ha for 27 FMUs, with the maximum size averaging between 53 and almost 12,500 ha (CCFM 2005). By comparison, clearcuts in 11 FMUs in the GLSL Region during this period averaged 5 to 130 ha, with a maximum size of 26 to 691 ha (CCFM 2005).

About 10% of the total area harvested annually is in the GLSL Forest Region where less intensive harvesting systems are used (83% of the total GLSL area) in which more than 40% of the overstory trees may be retained on site. The selection system is applied to the tolerant hardwood forests of the GLSL (~ 44% annual area harvested in GLSL) where single and group canopy openings formed by mortality due to wind, drought, insects, etc. are the most common natural disturbance (Frelich 2002). Stand-replacing natural disturbance of these forests by catastrophic wind storms is relatively infrequent, occurring once every 1000 years or more (Frelich

Table 5.1. Annual wood volume depletion (million m^3) due to harvesting and natural disturbance in Ontario's forest from 1996-2001 (Planning Area only). Wood volume losses associated with insects and diseases are due to mortality and growth reduction (OMNR 2002b).

Disturbance	Year 1996	1997	1998	1999	2000
Fire	41.6	1.8	7.3	12.8	0.4
Insects	10.6	7.7	5.8	4.9	5.9
Disease	12.9	12.9	14.5	12.9	13.3
Severe weather*	0.006	1.5	0.3	13.3	0.4
Harvest	21.4	22.2	21.3	22.6	24.7
Conifer	16.2	17.1	15.8	16.7	18.0
Hardwood	5.2	5.1	5.5	5.9	6.7
Total	**86.5**	**46.1**	**49.2**	**66.5**	**44.7**

* Volume depletion due to blowdown, drought, and cold weather damage.

2002). In the white and red pine forests of the GLSL, frequent low intensity surface fires (20 to 40 years) and stand-replacing fire at 200- to 300-year intervals are emulated using shelterwood and seed tree silvicultural systems (~ 40% annual area harvested) (Frelich 2002).

Harvesting may be accomplished by several methods that differ in the proportion of aboveground biomass removed from the site (OMNR 2003). Shortwood, or cut-to-length, logging is the least intensive form of harvesting and involves delimbing, topping, and cutting of tree boles to a desired length prior to their transport to landings. Tree-length logging refers to removal of only the merchantable portion of the bole, with delimbing and topping occurring at the stump, leaving branches and foliage on site. From 80 to 85% of the areas clearcut in Ontario are full-tree logged (D. Morris, 2005, OMNR, pers. comm.). Full-tree logging is the most intensive harvest method as it involves removal of the entire tree crown to roadside, where delimbing and topping occurs. This material may be returned to the cutover site, intact or as chips, but is most commonly piled and burned (Luke et al. 1993, OMNR 2001d). This results in removal of 15% to 100% more aboveground biomass, primarily the nutrient-rich fine branch and foliar tissue, than occurs with tree-length logging (Table 5.2) (Kimmins 1977, Freedman et al. 1981, 1986).

Harvesting, natural disturbance and forest carbon dynamics

A fundamental difference between the effects of harvesting and natural disturbance on forest C dynamics is the fate of biomass and DOM C pools after disturbance. Soil C remains relatively stable after any of these disturbances unless followed by an extreme treatment e.g., cultivation, slash burning, windrowing) (Johnson et al. 1992a, Mattson and Smith 1993, Freeden et al. 2005). However, more biomass C is generally removed and less DOM C remains on site after harvesting than natural disturbance (Krankina and Harmon 1994, Duvall and Grigal 1999, Thornley and Cannell 2000, McRae et al. 2001, Harmon and Marks 2002, Thornley and Cannell 2000). For example, clearcutting removes 70 to 90% of the standing biomass from the site and converts it into wood products. Depending on logging method, the tops, branches, and foliage may be left in the forest, immediately increasing the DOM C pool. For partial cut silvicultural systems, total forest C after harvest is comparatively high because a portion of stand biomass remains in residual overstory trees. Greater relative slash inputs to the DOM pool occur with partial cutting since cut-to-length and tree-length harvest are used to minimize damage to overstory trees and advance regeneration. Litter inputs from the residual overstory help minimize post-harvest reductions in the DOM pool. As well, the sheltering effect of a partial canopy on the understory environment may reduce decomposition and losses from the DOM C pool.

Natural disturbances result in comparatively little biomass loss and larger DOM inputs than harvesting (Krankina and Harmon 1994, Duvall and Grigal 1999, McRae et al. 2001). Fire typically consumes about 50% of understory vegetation and DOM, but as little as 20% of overstory stems and branches, fallen logs, etc. since this larger woody material burns only partially (Ohmann and Grigal 1979, Auclair 1985). The incompletely burned standing and fallen dead trees that remain provide large inputs to the post-disturbance DOM C pool. In contrast, insect infestation mortality occurs over several years, with large amounts of overstory biomass C gradually converted to DOM C pools (Gross 1985). Forest disturbance by disease, wind, drought, and other forms of severe weather may cause the abrupt or gradual mortality of a few or many overstory trees and their addition to the DOM C pool, but overall a large proportion of overstory biomass C is retained.

Biomass utilization and forest carbon dynamics

Full-tree harvesting removes a larger proportion of aboveground tree biomass than cut-to-length and tree-length logging methods. Extraction of the branches and foliage along with the stem removes nutrient-rich organic matter. Biomass utilization can affect forest C dynamics by influencing the size and composition of post-harvest DOM C pools, long-term site productivity, and the rate of stand regeneration.

Table 5.2. Estimated biomass C and nutrient removals for tree-length and full-tree harvest of mature stands of several forest types of eastern Canada. Number in bold for each forest type is percent additional biomass C or nutrient removal using full-tree relative to tree-length logging.

Forest type	Harvest method	Biomass C (t ha^{-1})	N	P	K (kg ha^{-1})	Ca	Mg
White birch[a]	Tree-length	52.8	125.7	8.6	50.8	126	23.2
	Full-tree	63.2	287.8	20.1	129.6	221.1	44.7
		19.7	**128.9**	**133.7**	**155.1**	**75.5**	**92.7**
Black spruce[b]	Tree-length	53.5	62	9	36	219	18
	Full-tree	71.0	234	18	122	334	43
		32.7	**277.4**	**100**	**238.8**	**52.5**	**138.9**
Red spruce-balsam fir[c]	Tree-length	52.6	95.2	16.3	56.4	117.6	16.5
	Full-tree	76.3	239.1	35.2	132.6	336.5	36.9
		45.1	**151.2**	**115.9**	**135.1**	**186.1**	**123.6**
Red pine-white pine-aspen[d]	Tree-length	68.8	146.0	19.8	88.5	301.3	37.1
	Full-tree	89.1	308.7	36.9	162.4	507.6	63.2
		29.5	**111.4**	**86.4**	**83.5**	**68.5**	**70.4**
Aspen[c]	Tree-length	20.8	66.1	7.1	52.7	108.8	15.3
	Full-tree	32.1	184.9	22.3	117.3	226.0	31.5
		54.3	**179.7**	**214.1**	**122.6**	**107.7**	**105.8**
White spruce[c]	Tree-length	52.0	94.4	15.6	53.7	175.2	18.9
	Full-tree	72.0	262.6	36.5	126	339.8	37.4
		38.5	**178.2**	**133.9**	**134.6**	**93.9**	**97.9**
Sugar maple-yellow birch[e]	Tree-length	77.2	246.5	13.7	148.4	589.0	27.9
	Full-tree	104.5	515.5	28.9	335.5	960.1	48.1
		35.4	**109.1**	**110.9**	**126.1**	**63.0**	**72.4**

Sources: [a]Titus et al. 1998, [b]Foster and Morrison 1987, [c]Freedman et al. 1986, [d]Hendrickson et al. 1987, [e]Morrison 1990

Biomass utilization effects on soil and dead organic matter carbon

Logging residues can be reduced greatly by full-tree harvesting (McInnis and Roberts 1995). Lower slash loads have raised concerns that soil C may decline as a result of reduced organic matter inputs (Hendrickson et al. 1989, Johnson 1992a). Lower slash loads could also reduce soil C by creating a forest floor microenvironment more favourable for soil respiration (Hendrickson et al. 1989). The long-term influence of biomass utilization intensity on soil C was examined using meta analysis of results from a number of field trials comparing full-tree with tree-length logging (Johnson and Curtis 2001). Harvesting method did affect soil C, with full-tree logging decreasing soil C by an average of 6%, and tree-length harvesting increasing soil C by 18% (Johnson and Curtis 2001). These and other results also suggest the effects of biomass utilization on soil C are more pronounced for coniferous species, with soil C in hardwood and mixedwoods relatively unaffected by biomass utilization (Knoepp and Swank 1997, Johnson and Todd 1998, Johnson and Curtis 2001, Johnson

et al. 2002). The influence of biomass utilization on soil C dynamics of northern coniferous forests is still poorly understood (Duckert and Morris 2001). However, the few existing field studies and model simulations indicate full-tree harvesting does not differ from other harvesting methods in effects on soil C (Bengtsson and Wikstrom 1993, Olsson et al. 1996).

Biomass utilization effects on site productivity and biomass carbon

More intensive biomass utilization removes larger amounts of organic matter and nutrients from the site than conventional logging methods (Kimmins 1977). This may influence physical, chemical, and biological properties of the soil that can affect site productivity (Johnson et al. 2002). Organic matter removals can reduce microbial biomass, non-symbiotic N fixation, mycorrhizal populations, and soil fauna in the short-term, all of which may affect site productivity (Entry et al. 1986, Jurgensen et al. 1997, Marshall 2000, Moore et al. 2002). The removal of nutrients in foliar and fine woody tissues may seriously reduce long-term site productivity depending on plant available nutrient pools and their natural replenishment by, for example, weathering, precipitation, dry deposition, mineralization, and biological N fixation (Weetman and Webber 1972, Kimmins 1977, Jurgensen et al. 1997). Early studies of the nutrient budgets of northern forest types indicated that additional nutrient losses of 15 to almost 400% could result from full-tree harvesting as compared with conventional, less intensive harvesting (Table 5.2) (Freedman et al. 1981, 1986). These studies indicated potential productivity loss due to nutrient depletion on poorer quality sites with low nutrient reserves and/or thinner soils (Boyle et al. 1973, Alban et al. 1978, Weetman and Algar 1983, Weetman and Webber 1972, Freedman et al. 1981, 1986, Hendrickson et al. 1987, 1989).

Little information is available on long-term growth response to more intensive biomass utilization (Egnell and Leijon 1999, Egnell and Valinger 2003). Planted Scots pine (*Pinus sylvestris* L.) and Norway spruce on N-limited sites in southern Sweden exhibited significant height, basal area, and stem volume growth reductions 24 years after full-tree as compared to tree-length harvesting (Egnell and Valinger 2003). The roughly 20% decrease in stem biomass production per tree resulted in a stand volume growth decrease of 0.8 $m^3\ ha^{-1}\ yr^{-1}$ over the study period. Depending on whether these growth reductions continue, the rotation length of full-tree harvested stands could be extended by 3 to 13 years. These results generally agree with studies comparing full-tree and tree-length harvesting used to commercially thin northern conifers in Scandanavia, and indicate an average growth reduction of the residual stand by 5 to 10% due to full-tree harvesting (Egnell and Leifon 1997 and references therein).

In general, the potential adverse effects of nutrient depletion and organic matter reduction by intensive biomass utilization is greater where rotations are shorter, on poorer sites, in cooler climates (shallow, coarse-textured soils), and where followed by intensive site preparation (Morris and Miller 1994, Worrell and Hampson 1997). In Ontario, current guidelines for emulating natural disturbance patterns discourage the use of full-tree logging on very shallow and very coarse-textured soils deemed potentially sensitive to nutrient losses associated with removal of nutrient-rich, fine woody debris (OMNR 2001d). Where full-tree harvesting is used, redistribution of logging slash from landings back to cutover is also encouraged (OMNR 2001d). These guidelines also recommend conserving organic matter as CWD by leaving unmerchantable logs on site and using cut-to-length and tree-length harvesting on sites deemed sensitive to nutrient depletion (OMNR 2001d).

Biomass utilization effects on forest regeneration and biomass carbon

Biomass utilization intensity can influence forest C by affecting the rate of stand regeneration (Section 2.2) (McInnis and Roberts 1995, Yanai et al. 2000, Waters et al. 2004). Tree-length logging damages the understory vegetation the least, and is therefore most appropriate where advance growth will provide the primary source of regeneration (Waters et al. 2004). Logging residue can also influence the establishment and growth of new regeneration through effects on seedbed, microenvironment, and growth of competing vegetation (Zabowski et al. 2000, Johnson et al. 2002). The removal of logging residues during full-tree harvest generally encourages

natural regeneration, particularly of shade intolerant species, by creating exposed mineral soil seedbed and a favourable microenvironment for establishment (Shearer and Schmidt 1999, Waters et al. 2004). However, competition by intolerant hardwoods and early successional species may be encouraged by this method, potentially reducing the regeneration of conifer species (Hendrickson 1988, Stevens and Hornung 1990, Emmett et al. 1991, McInnis anJ Roberts 1994). In contrast, higher slash loading after tree-length logging often results in lower surface soil temperatures, increased soil moisture contents, and lower competition, and tends to enhance regeneration of more shade-tolerant tree species (Smethurst and Nambiar 1990, Stevens and Hornung 1990, Proe and Dutch 1994, McInnis and Roberts 1995, Waters et al. 2004, Roberts et al. 2005).

Salvage logging to increase carbon storage

Salvage logging is increasingly used in North America to retrieve fibre from large, naturally disturbed areas (Fraser et al. 2004, Karr et al. 2004, Lindenmayer et al. 2004, Purdon et al. 2004). In extreme cases, such as occurred after 2000 in B.C., where the mountain pine beetle has damaged 4.2 million ha, salvage logging is planned for 4.9 million m^3 of timber (Bunnell et al. 2004). In the past, salvaging of burned areas was promoted in Ontario to retrieve usable fibre from damaged and undamaged residual trees. Current provincial efforts to emulate natural disturbance patterns when harvesting discourage salvage logging of burned areas, to retain fire-origin habitat on the landscape. Where salvage logging does occur, the same minimum residual standards (i.e., trees and snags ha^{-1}) that apply to clearcut harvesting are enforced (OMNR 2001d).

Currently, forest companies in Ontario seldom harvest dead and damaged trees from areas disturbed by fire, insects, or severe weather (OMNR 2004a). An average of 1.2 million m^3 yr^{-1}, about 5.4% of the total volume harvested annually from 1999 to 2003, was gathered by salvage logging, less than 1% of the total volume lost to disturbance each year. About 97% of the salvaged material comprised black and white spruce, jack pine, white pine, balsam fir and poplar, the same species that comprise 95% of the regular harvest. Salvaged trees were processed into structural materials (56%) and pulp and paper products (44%), with a small amount (0.2%) used for fuel. Salvaged fibre represented 1.9%, 9.7%, and 7.2% of the total volume harvested in the northeast, northwest, and southern regions of the province, respectively (OMNR 2004a). Kyoto Protocol accounting gives no credit for C stored in wood products from salvaged stands, even though the conversion of salvaged wood to forest products may result in longer-term C storage than leaving the dead trees to decompose in the forest.

Since salvaging must occur within a few years following damage to avoid fibre quality loss, it represents a second major disturbance to already disturbed forest ecosystems (Karr et al. 2004, Lindenmayer et al. 2004). Comparatively little is known of the ecological sustainability of this practice but some detrimental effects have been reported (Brais et al. 2000, Karr et al. 2004, Lindenmayer et al. 2004). For example, salvage logging of standing and fallen dead trees after wildfire may diminish the ecological benefits of burned areas that provide critical habitat to many insect, animal, and plant species (Freedman et al. 1996, Morrisette et al. 2002, Bunnell et al. 2004, Nappi et al. 2004, Hanson and Stuart 2005). Salvaging standing trees killed by wildfire can alter vegetation dynamics by removing the aerial seed bank of species with serotinous cones and increasing insolation of the forest floor, inhibiting the regeneration of some fire-adapted plant species (Martinez-Sanchez et al. 1999, Fraser et al. 2004, Purdon et al. 2004). The soils of burned areas are quite sensitive to additional disturbance by harvesting and are easily compacted and eroded (Karr et al. 2004, Lindenmayer et al. 2004). Until the ecological effects of this practice are better understood, conventional harvesting methods should be modified in salvage operations to minimize the potential adverse effects on sensitive components of terrestrial and aquatic ecosystems (Brais et al. 2000, Bunnell et al. 2004, Karr et al. 2004, Lindenmayer et al. 2004, Nappi et al. 2004, Purdon et al. 2004).

Harvest frequency and forest carbon storage

The rotation, or the number of years between stand establishment and final harvest of managed forests, is determined by the time required to achieve a desired size or stage of biological or economic maturity. By comparison, the natural ecological rotation of unmanaged forests is determined by the frequency of stand-replacing disturbance or the period of natural succession where disturbance does not occur. Because forests accumulate C beyond the age of maximum sustained yield or financial return, managed forests may contain 50% to 25% less C than natural forests (Cooper 1983, Harmon et al. 1990, Karjalainen 1996). Most of the additional C of unmanaged forests is in the form of dead wood that increases the DOM C pool with stand age (Krankina and Harmon 1994, Fleming and Freedman 1998, McRae et al. 2001). Natural, unmanaged forests have consistently been shown to contain more C than their managed counterparts. The largest potential losses in forest C occur when old-growth stands of long-lived forest types are harvested and converted to intensively managed plantations with a rotation much shorter than the natural disturbance interval (Harmon et al. 1990, Harmon and Marks 2002, Janisch and Harmon 2002). For example, the harvest of 450-year-old Douglas-fir–western hemlock (*Tsuga heterophylla* (Raf.) Sarg.) forests of coastal regions of the Pacific Northwest of North America and their conversion to Douglas-fir plantations managed on a 60-year rotation can reduce C storage by 220 to 230% (Harmon et al. 1990).

Simulations demonstrate that longer rotations can increase C storage in managed forests (Aber et al. 1978, Cooper 1983, Cannell 1996, Karjalainen 1996, Dewar and Cannell 1992, Thornley and Cannell 2000, Liski et al. 2001, Euskirchen et al. 2002, Harmon and Marks 2002). Harvest frequency has also been shown to be more important to forest C storage than harvest intensity, biomass utilization, and the amount of slash left on site (Aber et al. 1978, Dewar and Cannell 1992, Harmon and Marks 2002). Generally, for a given rotation length, C storage can be increased by removing a smaller fraction of trees, a lower proportion of their aboveground biomass, and retaining more slash on site.

Maximizing forest C storage can be achieved by a landscape dominated by old-growth forests or where harvesting occurs at or beyond the age of maximum NEP (Schulze et al. 2000, Euskirchen et al. 2002). However, reducing harvest frequency and area without compensating actions to increase timber supply will decrease the amount of wood available to the forest industry. Obviously, some balance between C storage and wood supply is needed to assure that public demand for forest products can be met. Harmon and Marks (2002) examined rotation lengths of 40 to 120 years, in combination with different harvest intensity and biomass utilization, and logging residue retention on forest C storage and wood supply for coastal Douglas-fir forests in the Pacific Northwest. The best compromise between C storage and wood production was achieved by 80- to 100-year rotations, comparatively low harvest intensity and biomass utilization (e.g., partial cutting, tree-length harvesting, protection of advance growth), and leaving logging slash on site. Similarly, Thornley and Cannell (2000) concluded that the optimal balance of wood yield and C storage is attained using harvesting systems that retain some forest cover and mimic natural disturbance regimes. Use of more intensive forest management practices can also offset losses in wood supply that would occur with a reduction in the annual area harvested. The magnitude of the potential increases in wood supply and forest C that can be achieved by keeping rotation length the same but increasing tree biomass growth rates through intensified management are detailed throughout this report and elsewhere (Krankina and Harmon 1994, Binkley et al. 1997, Parker et al. 2000, Thornley and Cannell 2000).

Reducing harvest frequency in Ontario to increase forest carbon

The amount of C potentially stored in managed forests by reduced frequency of harvest depends on the natural disturbance cycle (Price et al. 1996, Harmon and Marks 2002). Extended rotations of boreal forest species to increase C storage can only be achieved at the cost of fire suppression and protection from insect

outbreaks, which may be impractical for several reasons (Section 4.0). As well, most of Ontario's boreal forests support short-lived species, such as jack pine and aspen, and may not accumulate as much C as other forest types. Focusing a C sequestration strategy of longer rotations on less productive sites and slower growing species, such as lowland black spruce forests, would produce a relatively small benefit compared to the greater C sequestration rates of faster growing species on better sites (Dewar and Cannell 1992). For these reasons, lengthening rotations to increase forest C storage in Ontario may be more practical for tolerant hardwoods or white and red pine forests that have much longer natural disturbance intervals.

Despite these potential limitations to increasing forest C storage by reduced harvest frequency, the effect of longer rotations on forest C can be demonstrated from results of model simulations using CBM-CFS2. Reducing the 200,000 ha harvested annually by 1,000 ha (a reduction in area harvested of 0.5%) would be roughly equivalent to about 20 ha less area harvested in each of the province's 47 FMUs. An average 100-year-old black spruce stand contains 38 t ha^{-1} of merchantable biomass C. Therefore, a 0.5% reduction in harvest area would preserve 1,000 ha of forest and would retain about 38,000 t of C. Disregarding differences in biomass C accumulation between harvested and regenerating stands, reducing harvest by 1,000 ha yr^{-1} annually from 2008 through 2012 would conserve a total of 190,000 t of biomass C, provided the stands are not killed by fire or another natural disturbance.

6. The Potential Contributions of Forest Management to Carbon Storage in Ontario's Forests: A Summary

6. The Potential Contributions of Forest Management to Carbon Storage in Ontario's Forests: A Summary

Canada potentially will achieve its target for GHG reductions agreed to in the Kyoto Protocol through some combination of reduced fossil fuel burning and the planned sequestration of C in biological sinks, including forests. For this reason, there is interest in the amount of additional C that can be stored in trees, forest soils, and long-lived wood products by altering how forest management is practiced. Canada must decide by 2006 whether to include forest management in its Kyoto forest C accounting framework. If forest management is included in Canada's Kyoto accounting, the country will be responsible for both increases and decreases in C storage.

Any increase in forest area creates a sink for CO_2 and therefore helps reduce the continued accumulation of GHGs in the atmosphere. Under the Kyoto Protocol, two articles are especially relevant to forest C storage. Article 3.3 gives credit for C sequestered by the creation of new forests by afforestation since 1990. Debits, meanwhile, accumulate due to C released as a result of deforestation since 1990. Further, to be claimed as credits, afforestation and reforestation must result from human intervention, such as tree planting or seeding, rather than from natural conversions to forest.

Article 3.4 of the Kyoto Protocol addresses C credits resulting from forest management. Article 3.4 allows the managed forest C sink to offset forest C losses resulting from the net effects of deforestation and afforestation, defined in Article 3.3. In the Kyoto Protocol, Canada agreed to a cap of 9 Mt C yr^{-1} from 2008 to 2012 from forest management to offset a C debit due to deforestation. If the above condition is met, then Article 3.4 allows an additional credit for forest management of up to 12 Mt of C yr^{-1} (for a total C credit of up to 21 Mt yr^{-1}).

Changes in forest C are the net result of increased biomass and DOM C due to tree growth and litter accumulation, and reduced C caused by natural disturbance, harvesting and organic matter decomposition. If during the Kyoto accounting period (2008 to 2012) the increase in forest C due to tree growth is greater than the decrease caused by fire, insects, harvesting, and decomposition, then it will be possible to claim some or all of the 12 Mt annual credit due to forest management. Realization of these forest C credits depends on the level of natural disturbance. An extreme fire year or a major outbreak by spruce budworm during the Kyoto accounting period would reduce the amount of C stored in Ontario's forests and, together with C releases attributable to harvesting, could make Ontario a net source of C. However, should Ontario be a net source of C, so long as the forests in Canada as a whole functioned as a net C sink, then Canada would qualify to use the forest C credits accrued from forest management. How forest C credits or debits under Article 3.4 would be distributed has yet to be announced.

The use of more intensive forest management practices in Ontario's forests can increase C storage and wood yield. This opportunity exists because, beyond fire protection, relatively little forest area is intensively managed. North of the Planning Area, almost all forest area is unmanaged (fuel wood harvest by remote communities on 1% of the area being the only forestry activity). Within the Planning Area, it has been estimated that 36% of the area is unmanaged, 32% is managed extensively, 30% receives basic forest management, and only 2% receives intensive forest management (Section 1.3) (W. Bell, 2005, OMNR, pers. comm.). Therefore, most of Ontario's forests (68% of the Planning Area) do not receive any silvicultural intervention to secure either natural or artificial regeneration. Unassisted natural regeneration precludes selection of species composition of the future stand as well as foregoing the gains in C and wood supply that can be obtained through planting (Section 2.2). In certain cases, the lack of basic management to encourage regeneration of desired species may lead to an extended period of slow regeneration and/or understocked stands. While an average of 112,000 ha have been planted and seeded annually since 1990, other stand establishment practices affecting C storage,

such as site preparation (Section 2.1) and vegetation management (Section 2.3), are applied to only about one-third of disturbed areas (Figure 6.1). More intensive forest management practices, such as planting genetically improved stock (Section 3.1), stand density regulation (Section 3.2), forest fertilization (Section 3.3), control of insect pests (Section 4.2), and the rehabilitation of roads and landings (Section 5.1) are either applied on a very small scale (CCFM 2005) or are not currently approved for operational use (OEAB 1994).

Increasing the area regenerated by planting

Planting increases the rate at which forest C accumulates in biomass and DOM, assuming adequate site preparation and vegetation management treatments are also applied (Section 2.0). Many areas of the boreal forest regenerate rapidly to fire-adapted pioneer species, such as aspen, white birch or jack pine, without silvicultural intervention and if these species are acceptable growing stock there would be little benefit to planting such sites. However, planting would increase C storage and wood supply where growth of natural regeneration is slow or where stocking of preferred species is less than desired.

On average, almost 73,000 ha were planted annually in Ontario from 1990 to 2000. Based on this average rate of planting, the total C gain in Ontario by 2012 due to planting beginning in 1990 will be about 16 Mt. Extrapolating to 2070, the gain in C from planting from 1990 to 2012 would total about 28 Mt. Viewed over the longer-term, for every hectare planted the increase in total stand C would be about 16.5 t as early as 15 years and as long as 80 years after planting. If an additional 10,000 ha were planted in 2005, the gain in total C storage would level off at approximately 165,000 t for the period from 2022 to 2070. If the area planted was increased by 10,000 ha over the current level to 83,000 ha yr^{-1} for the next 50 years beginning in 2005, the total gain in C (with the assumption of no stand losses from natural disturbance) from this 14% increase in annual area planted would be about 8.25 Mt.

Using genetically improved planting stock

Gains in tree growth and forest C storage can be achieved by planting genetically improved progeny from seed orchards (Section 3.1). Conservatively, growth gains of 5% are achievable through genetic improvement if planted stands are established with vigorous nursery stock on prepared and tended sites. Planting faster growing trees reduces the age at which maximum biomass is achieved, but *the maximum amount of C that can be stored in living biomass is the same as in stands of non-improved trees.* If improved and non-improved stands are harvested at equivalent merchantable volumes, then the gain in biomass C from genetic improvement

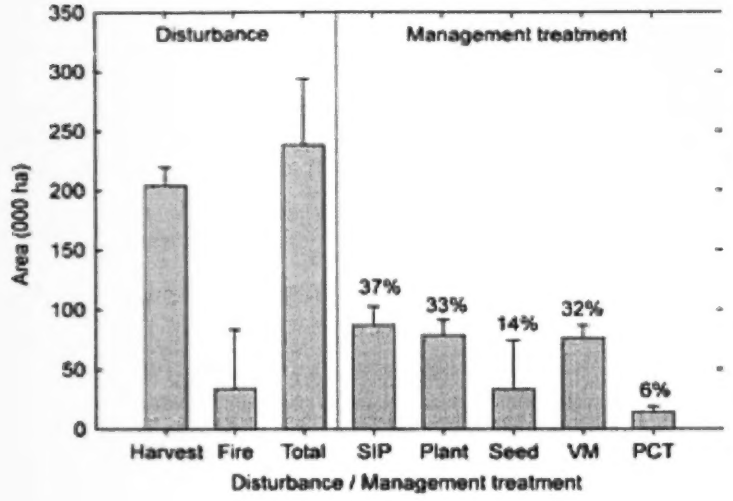

Figure 6.1.
Average productive forest area disturbed by harvesting and fire compared to area receiving several forest management treatments in Ontario from 1990-2003. Numbers above vertical bars are percentage of total disturbed area receiving a given management treatment (CCFM 2005); SIP – site preparation, VM – vegetation management, PCT – pre-commercial thinning. Some areas receive more than one management treatment.

is related to differences in rotation lengths. A black spruce Site Class 2 plantation established using genetically improved planting stock will attain biomass C equivalent to that of an 80-year-old naturally regenerated stand 13 years earlier than the naturally regenerated stand. The reduction in rotation of improved plantations is 3 years compared to plantations of non-improved stock. Eighty years after stand establishment, the gain in biomass C storage for improved plantations would be about 8 t ha^{-1} compared to natural regeneration, and less than 1 t ha^{-1} compared to plantations of non-improved trees.

Assuming the area of genetically improved plantations increased from 0% to 15% of provincial planting from 1996 to 2000, was 20% in 2001, and increased by 5% per year thereafter, the gain in biomass C in 2012 from all genetically improved stands planted since 1990 compared to stands of non-improved stock would be over 76,000 t. Compared to naturally regenerated stands, the total gain in C between 1990 and 2012 from planting genetically improved trees would be almost 2.2 Mt.

Lengthening natural disturbance cycles through forest protection

One of the most effective ways to increase forest C storage is to lengthen the natural disturbance cycle. Stand-replacing disturbances initiate a cycle in which biomass and DOM C pools initially decline and take decades to begin accumulating C again. Longer disturbance cycles keep a larger proportion of older stands in the mature growth phase that contain more C in biomass and DOM. Some stand-replacing disturbances occur as a result of extreme environmental conditions, such as high winds and ice storms, and their timing and area affected cannot be managed by human intervention. In comparison, fire and insects are natural disturbances that to some extent can be controlled through forest management.

Fire management

Wildfire is the dominant stand-replacing natural disturbance agent for many of Ontario's forest types, with an average of 1,480 fires burning 140,000 ha yr^{-1} from 1917 to 2000 (Section 4.1). Most of the immediate effect of fire on forest C is from direct C losses to combustion, with 40 to 60% of the C in branches and foliage in living vegetation and the forest floor typically being consumed. Larger woody material in standing living and dead trees, and fallen stems in the forest floor, are usually only partially burned during a fire. However, it is the post-fire C losses to decomposition that account for the largest effects of fire on forest C. Stands are generally a net source of C for several decades after fire, as decomposition greatly exceeds C uptake by vegetation. As forests grow and reach maturity, they accumulate large amounts of DOM C. Reducing fire frequency therefore increases the amount of forest C on the landscape both by decreasing the area of recently burned forest (C sources) and increasing the number of older stands (C sinks).

Fire suppression has increased the forest area occupied by older stand age classes with greater C stocks (Martell 1994, Ward et al. 2001). Further increasing the intensity of fire management to reduce the area burned can increase the area of older forests, and therefore the total amount of forest C on the landscape. For example, from 1990 to 2001, intensive fire management resulted in 60% less area burned compared to that burned where a measured response was applied. Assuming 'average' fire conditions occur, expanding intensive fire management to include an additional 11 million ha in the northern boreal forest for the period from 2008 to 2012 would retain an additional 4.1 million t of stored forest C on the landscape through decreased area burned.

Controlling insect outbreaks

Insects and disease are also significant natural disturbance agents in Ontario's forests that cause growth reductions and tree mortality (Section 4.1). Defoliating insects alone have affected from 8 to 40 million ha yr^{-1} from 1975 to 2000. Annual losses in biomass C due to insect pests and disease in Ontario from 1994 to 1999 ranged from about 8.6 M t to 11.3 Mt. Spruce budworm, which was not at outbreak levels during the 1990s, was responsible for the largest losses of biomass C during this period, ranging from about 2.1 to 6.2 Mt yr^{-1}. Given that a major outbreak of spruce budworm is anticipated within the next decade, current biomass C losses are expected to increase dramatically.

Although spraying insecticides can prevent stand mortality during a spruce budworm outbreak, it would not greatly increase C storage in Ontario. Most of the mortality following repeated defoliation occurs in balsam fir (Gross 1985), with damage to more economically valuable black and white spruce being much lower. Use of insecticides in Ontario's forest is done mainly to protect high value, commercially operable (i.e., accessible) stands in insect outbreak epicenters or for containment of outbreaks. Therefore, spraying would in most cases be done to preserve older, spruce-dominated stands for harvest. Since harvest removes more C from the site than does a spruce budworm infestation, spraying would indirectly reduce forest C storage compared to allowing the outbreak cycles to progress naturally.

Budworm control could increase forest C storage by reducing growth losses associated with defoliation. Repeated moderate to heavy spruce budworm defoliation causes growth losses of 22% after one year to as much as 78% after seven years in black and white spruce (Gross 1985). Maximum growth rates for Site Class 2 black spruce stands occur at 40 to 60 years of age, when approximately 1 t of biomass C ha^{-1} accumulates annually. Through five successive years of heavy spruce budworm infestation, biomass C accumulation declines from about 1.0 t of C ha^{-1} to 0.4 t of C ha^{-1} by year 5 (Gross 1985). If by spraying a black spruce stand once, heavy defoliation by spruce budworm was prevented for five years, then the additional biomass C sequestered would be about 2.0 t ha^{-1} compared to severely defoliated stands.

In the 1990s during a moderate infestation, 60,000 ha were treated to control spruce budworm in Ontario. A similar 60,000 ha spray program in 2008 would result in a C gain of about 122,000 t by 2012. Spraying a further 60,000 ha in 2009 would increase forest C storage by about 89,000 t by 2012, with an additional C gain of 56,000, 32,000, and 13,000 t resulting from treatment of 60,000 ha in 2010, 2011, and 2012, respectively. Cumulatively, protecting 60,000 ha of black spruce from moderate to heavy defoliation by spruce budworm each year would increase forest C by 311,000 t by 2012 relative to allowing the outbreak to continue unimpeded. However, if after conserving this additional 311,000 t of C by spraying, the stands were harvested, much of the gain in C storage would be negated because merchantable timber removed by harvest is considered a C loss under Kyoto rules. Thus, the benefit in C storage resulting from spraying depends on whether the protected stands are harvested.

Minimizing harvest-related disturbance

Similar to the effects of stand-replacing natural disturbances on forest C, harvesting results in a transient loss of forest C that may last for decades. However, harvesting is a human disturbance in which the timing, intensity, and location of its occurrence can be regulated to favour C storage. As well, minimizing additional site disturbance by heavy equipment used for harvesting, and the temporary or permanent loss of productive forest area to roads and landings can reduce forest C losses associated with timber extraction.

Managing skid trails

While skid trails do not result in deforestation, the compaction of fine-textured soils by skidders can reduce tree growth and forest C storage (Section 5.1). Conversely, on dry sites with coarse-textured soils where low soil moisture can limit growth, compaction can improve tree growth by increasing soil water retention. There are no guidelines specifically governing the area of skid trails permitted on harvested areas, although provincial guidelines for avoiding soil compaction exist (Archibald et al. 1997). Rather, skid trail density depends on the layout of roads and landings, the discretion of the skidder operator, and the topography of the site. McNabb and Froelich (1984) estimate that the area affected by skid trail compaction during summer harvesting is at least 10% of the area harvested (about 20,000 ha yr^{-1} in Ontario). While natural processes can reverse skid trail compaction, without some form of rehabilitative treatment, which is not required in Ontario, this can take decades. Statistics are not available to predict the benefit in C sequestration that might be derived from better tree growth resulting from more closely regulated skid trail layout. However, growth and C gains will likely result from reducing the area of skid trails, particularly on sensitive sites.

Minimizing area of roads and landings

Forest access roads average about 4% of the total stand area harvested (Section 5.1) and are classified by their intended length of use: Class 1 (permanent roads), Class 2 (used for up to 15 years) or Class 3 (not scheduled for use beyond the harvest). After harvest, Class 3 roads are decommissioned, which in Ontario means they are no longer used and may be replanted. However, with the exception of winter roads, trees on decommissioned roads grow slowly due to compaction and removal of organic matter and topsoil. Accordingly, active and decommissioned forest access roads could be counted as deforested land. Landings similarly add to deforestation and account for about 2% of the harvested area annually.

In the approximately 200,000 ha harvested each year since 1990, we assume that the construction of new roads and landings deforested 2% of the area, or about 4,000 ha yr^{-1}. The road and right-of-ways do not regenerate in a timely manner to forest, and consequently do not contribute to biomass C accumulation. Over time, this causes a loss of C on areas converted to roads and landings compared to forest that was harvested and regenerated by planting (see Section 5.1, Fig. 5.1).

Areas deforested for roads or landings in 1990 were estimated to contain about 25 t less C ha^{-1} in 2012 than forest regenerated by planting after clearcut harvesting in 1990. Multiplied by the approximately 4,000 ha assumed to have been converted to roads and landings just in 1990, total C loss would be about 102,000 t in 2012. Total deforestation due to roads and landings from 1990 to 2012 would result in a loss of about 92,000 ha of productive forest and 960,000 t less forest C compared to replanted forest. Reducing the area of new roads and landings by 10% beginning in 2005 would increase C storage in Ontario's forests by about 9,000 t in 2012.

Extending forest management into virgin forest north of the Planning Area will also result in deforestation of 4% of the harvested area due to the construction of new access roads and landings. If 10,000 ha of the 4.5 million ha of this natural forest were harvested each year from 2005 to 2011, about 9,000 t of C would be lost to road construction by 2012. By 2070, forest C on area occupied by roads and landings constructed in 2005 would be reduced by about 93 t ha^{-1}. Thus, on the 400 ha of roads and landings constructed in 2005, the reduction in C storage would be about 37,372 t. Harvesting 10,000 ha of new forest annually from 2005 to 2055 (a total harvest of 500,000 ha over 50 years), would convert 20,000 ha of forest to roads and landings. By 2070, this would decrease C storage by about 1.1 Mt. Lowering the rate of deforestation due to roads and landings by 10% (from 400 to 360 ha yr^{-1}) would reduce the loss of C in 2070 by about 100,000 t. Alternatively, rehabilitation of roads and landings, particularly on high productivity sites, should be considered as a means to reduce deforestation and increase forest area and C storage.

Reducing harvest frequency

Reducing harvest frequency has a relatively large positive effect on forest C storage compared with decreasing harvesting intensity (clearcut versus partial cut) or biomass utilization (full-tree versus tree-length harvest) (Section 5.2). Use of longer rotations for certain forest types would increase the area of older stands that contain large amounts of biomass and DOM C on the landscape at a given time. If the annual harvest was reduced by 1,000 ha (producing a 0.5% reduction in harvest over the approximately 200,000 ha harvested each year), this would reduce the harvest in each of the province's 47 FMUs by slightly more than 20 ha. If the average harvested black spruce stand is 100 years old, then 38 t ha^{-1} merchantable biomass C would be removed from the sites by harvest. Therefore, a 0.5% reduction in harvest area would retain about 38,000 t of C. Reducing harvest by 1,000 ha yr^{-1} from 2008 to 2012 would conserve a total of 190,000 t of biomass C. Reducing the area harvested would contribute to making Ontario's forests a net GCG sink. In the short-term, the area of disturbed forest could be reduced by increasing the use of salvage harvest after fire or insect outbreaks (Section 5.2). This effectively lengthens the disturbance cycle without reducing the supply of wood fibre for forest industry.

Conclusions and Future Directions

This report examines 10 forest management practices that affect the C balance of forests. These 10 activities fall into four categories: stand establishment (site preparation, planting/seeding, competition control), growth enhancement (genetic improvement, thinning, fertilization), forest protection (from forest fires, insects, disease), and harvest (controlling the area of roads, skid trails, landings; controlling the intensity and frequency of harvest; controlling utilization). All of the practices, except thinning, can increase forest C storage by either increasing the rate of C accumulation or by protecting or retaining forest C already present. Only fertilization can increase the inherent productivity of a site and the maximum potential amount of C that can be stored, but is not an approved forest management practice in Ontario. A general indication of the present frequency of use of these practices in Ontario is provided in Table 6.1. This could be used as a base to measure future progress.

The forest management activities considered in this report will result in relatively small gains in C between 2008 and 2012. For example, prior to 1997 the planting of genetically improved trees was virtually non-existent. Because the rate of C sequestration by trees is relatively slow in the years shortly after stand establishment, the gain in C from genetically improved stands from 1990 to 2012 would be about 76,000 t. Of this total, perhaps 30,000 t C would accrue for credit in the 2008 to 2012 accounting period. By comparison, the effects of planting non-improved trees will result in considerable increases in forest C. Planting since 1990 is estimated to increase total forest C by 16 Mt in 2012. The average annual increase in forest C between 2008 and 2012 for planting in Ontario from 1990 through 2006 is about 0.8 t ha^{-1}. Planting will result in the sequestration of about 880,000 t additional C in each year of the 2008 to 2012 accounting period. Large amounts of C will be sequestered as a result of practices such as fire suppression and control of spruce budworm outbreaks.

A significant change in forest C storage can reflect the sustainability of forest management. Increasing C storage in forests not only mitigates atmospheric GHG accumulation and global climate change, but also reflects increased potential for forests to support economically, biologically, and socially important activities. Although not all of these practices will produce large increases in forest C in the first Kyoto accounting period, the practices examined in this report will in almost all cases increase both the rate of C storage and provincial wood supply. Using these practices appropriately will likely yield the highest long-term economic benefits to Ontario and its forest sector.

There is a need for improved understanding of C sink/source relationships in forests. The values of C storage estimated in this report are coarse estimates based on simplified assumptions about forest productivity. An estimate of the gains in forest C sequestration can be achieved using the Strategic Forest Management Model (SFMM) and associated forest resources inventory information. To address this need, a large-scale forest C modelling project was initiated in 2002. The background and objectives of this project are described by Boivin et al. (2005). This large-scale forest C project will use C models to track the flow of C in Ontario forests.

Table 6.1. Frequency of use of various forest management practices that increase C sequestration in boreal conifer (C) and intolerant hardwood (H) forests of Ontario. Frequency of use is based on the area treated: rarely <5%; seldom ≥5% to <25%; occasionally ≥25% to ≤75%; often >75%.

Forest management activity	Rarely	Seldom	Occasionally	Often	Always
1. Stand establishment					
Site preparation	H		C		
Planting	H		C		
Seeding	H	C			
Vegetation management	H		C		
2. Growth enhancement					
Planting improved stock	H	C			
Pre-commercial thinning	H	C			
Commercial thinning	N/A*				
Fertilization	N/A*				
3. Forest protection					
Fire management					C, H
Insect control	H	C			
4. Harvesting					
Preharvest planning of roads and skid trails to reduce total area	H	C			
Road and landing rehabilitation	C, H				

*Not currently approved for use in Ontario.

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